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## POTENTIAL FOR BIOMAGNIFICATION OF CONTAMINANTS WITHIN MARINE AND FRESHWATER FOOD WEBS

by

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dieldrin, endrin, kepone, mirex, atrazine, endosulfan, lindane, chlorinated phenols, chlorinated benzenes, and polynuclear aromatic hydrocarbons (PAH). The few contaminants which appeared to have the potential to biomagnify in aquatic systems included methylmercury, PCB, possibly kepone and mirex, and some PAH's, including benzo[a]pyrene and naphthalenes. Based upon the information reviewed, it may be concluded that the biomagnification of contaminants in marine and freshwater food webs is not a dramatic phenomenon. As the biological availability of contaminants from sediments should be similar regardless of whether or not these sediments have been dredged and placed in an open-water disposal site, it is unlikely that open-water disposal of contaminated dredged material will cause any widespread perturbations in marine or freshwater ecosystems. Existing data gaps were identified, and recommendations were made for guiding research efforts on the question of contaminant biomagnification in aquatic food webs.

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## EXECUTIVE SUMMARY

The total yearly volume of materials dredged by combined Corps of Engineers and private operations averages about 290,000,000 m<sup>3</sup>. Pesticides and pesticide residues, nutrients, organic wastes, heavy metals, and other contaminants entering our waterways may associate strongly with particulate materials and eventually accumulate in the sediments. The presence of high levels of potentially toxic contaminants in some sediments has generated concern that dredging operations and the disposal of dredged material may cause the deterioration of the environment. Chemical residues which persist in the environment may be absorbed by plants and animals and accumulate within their tissues to levels that are greatly in excess of the ambient concentrations in their environment. Many of these substances have no known biological function and could accumulate to levels that are detrimental to the organism itself, or to its predators. Biomagnification may occur if the contaminant is persistent in biological systems and the food pathway is essentially linear, with the predominant energy flow from lower to higher trophic levels. (The meanings of biomagnification, bioaccumulation, and bioconcentration are defined as used in this text.)

Although well known in terrestrial ecosystems, the occurrence of biomagnification in aquatic ecosystems is questionable and is the topic of considerable debate. The objectives of this report are multifold: review the literature on biomagnification of contaminants within aquatic ecosystems; determine the relative importance of food as a source of contaminants in aquatic food webs; pinpoint those contaminants which may significantly biomagnify within aquatic food webs; indicate the gaps in existing knowledge; and provide recommendations for future research on biomagnification of contaminants in aquatic systems. This report is part of a study to assess the potential impact of the open-water disposal of contaminated dredged material upon aquatic ecosystems and is limited in scope to water-breathing aquatic animals.

The literature treating the bioconcentration of contaminants by and the toxicity of contaminants to marine and freshwater organisms is voluminous, in contrast to that regarding biomagnification. The available information suggests that mercury, particularly methylmercury, may be the only heavy metal that biomagnifies significantly within aquatic food webs. Food is also an important source of copper, zinc, and selenium, all of which are essential

trace elements for animal metabolism, as well as arsenic, chromium, lead, and possibly cadmium, which are not known to have any biological functions. These metals do not biomagnify, however. Organic compounds which appear to have significant potential for biomagnification include polychlorinated biphenyls (PCBs), benzo[a]pyrene, the naphthalenes, and, possibly, a few organochlorine insecticides, such as dieldrin, endrin, kepone, and mirex. Relatively little food-chain information was available for other organic compounds, however. The data available indicate that biomagnification of contaminants in freshwater and marine food webs is not a dramatic phenomenon. As the biological availability of contaminants from sediments should be similar regardless of whether or not these sediments have been dredged and placed in an open-water disposal site, it appears unlikely that the open-water disposal of dredged material will have any substantial environmental impacts.

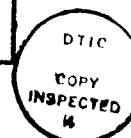
Several important ideas regarding future research efforts have surfaced in this review and will now be summarized briefly. More emphasis needs to be placed upon using the proper experimental design to address the problem and upon using adequate numbers of experimental organisms to account for natural variation in the population. The concentration of a contaminant within living organisms should be expressed in as many ways as possible (fresh weight, dry weight, tissue, organ, lipid, etc.) to allow valid comparison with work done elsewhere. For the purpose of biomagnification studies specifically, the expression of contaminant concentrations on the basis of parts per million dry weight of the whole organism (with and without gut contents, where possible) is the most useful approach.

From the perspective of field-oriented research, a number of recommendations have emerged. Trophic levels must be precisely determined using an accepted method, rather than by arbitrary assignment. When sampling in the field, all possible trophic levels should be collected at a given place and date, with a regular sampling schedule. Information on an organism's size, age, sex, and physiological state should be recorded, if possible. Gut contents should be analyzed chemically, and for species composition wherever possible. Data on physicochemical conditions should be taken at each place and on each date. Data from on-going field studies should be compared with those from any previous work at the same location.

Laboratory studies need to concentrate upon those compounds which have very low water solubilities and high solubilities within specific tissue

fractions, particularly fats or lipids. Chronic exposure to contaminants should be done at levels approximating those in nature and without the use of solvents, carriers, or chelators which may enhance water solubility and biological availability. Experimental food chains should include only species that actually are representative of those found in the natural ecosystem. Environmental conditions during exposure must also reflect as closely as possible those actually occurring in the natural ecosystem, or the data will have no valid application to a real system. Both the chemical species of a contaminant encountered in nature and its depuration following exposure must be considered. Background levels of the contaminant in the experimental organism and the organism's possible requirement for the contaminant (in the case of essential metals) must be evaluated in any bioaccumulation study. Finally, when using radioisotopes to follow the movement of a contaminant in a food chain, the data should be presented in terms of absolute concentration (parts per million, etc.), as well as in radiological terms such as disintegrations per minute.

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## PREFACE

This review was prepared at the US Army Engineer Waterways Experiment Station (WES), Environmental Laboratory (EL), during the period February to August 1983 by Dr. Stratford H. Kay of the Contaminant Mobility Research Team. The report was under the general supervision of Dr. R. M. Engler, Chief, Contaminant Mobility and Regulatory Criteria Group; Mr. D. L. Robey, Chief, Ecosystem Research and Simulation Division (ERSD); and Dr. J. Harrison, Chief, EL. Assistance and technical comments were also received from other members of the ERSD. This study was conducted as part of the Long-Term Effects of Dredging Operations (LEDO) Program. The LEDO Program is managed through the Environmental Effects of Dredging Programs, Mr. C. C. Calhoun, Jr., Manager. The LEDO Technical Monitors are Drs. W. L. Klesch and R. J. Pierce, Office, Chief of Engineers (OCE), and Mr. Charles W. Hummer, Dredging Division, Water Resources Support Center.

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POTENTIAL FOR BIOMAGNIFICATION OF CONTAMINANTS WITHIN  
MARINE AND FRESHWATER FOOD WEBS

PART I: INTRODUCTION

Background

1. Annually the US Army Corps of Engineers (CE) dredges more than 200,000,000 m<sup>3</sup> of sediments to maintain navigable channels in the waterways of the United States (U.S. Army Engineer Water Resources Support Center 1979). The total yearly volume of materials dredged by combined CE and private operations averages about 290,000,000 m<sup>3</sup> in maintenance operations and 78,000,000 m<sup>3</sup> in new work dredging operations (Engler 1980). Pesticides and pesticide residues, nutrients, organic wastes, heavy metals, and other contaminants enter our waterways from many sources, including drainage from the land, mining operations, and waste disposal. Many of these contaminants associate strongly with particulate materials and eventually accumulate in the sediments. Disposal of dredged material in open water generally is convenient and relatively inexpensive and, consequently, is used extensively by the CE (Center for Wetland Resources 1977). The presence of potentially toxic contaminants in some sediments (US Environmental Protection Agency 1975a, 1975b; Johanson and Johnson 1976) has generated concern that dredging operations and the disposal of dredged material may cause the deterioration of the environment. The underlying basis for this concern stems from public health considerations regarding human consumption of contaminated foods and the possible environmental perturbations caused by contaminants.

2. The fate of contaminants in the environment depends upon a variety of factors, including the chemical and physical properties of the specific contaminant, its residues (degradation products), and its metabolic by-products, as well as the characteristics of the sediments with which the contaminants are associated. Chemical residues which persist in the environment may be accumulated by plants and animals and, thus, may enter the food web. Once taken up by a plant or animal, a chemical residue may have any of several fates: it may be accumulated and stored in one or more tissues or organs; it may be eliminated actively or passively from the organism; or it may be metabolized and its residues either stored or eliminated. The influence of a specific

environmental contaminant on the biota depends upon both the contaminant and the organism of concern, and may range from no apparent effect to chronic toxicity (long-term effect) to acute toxicity (short-term effect). The scientific literature contains numerous studies on the effects of various chemicals upon living organisms, and many volumes have been published reviewing this subject.

3. Many chemicals are frequently present in the environment in extremely low concentrations, often below the levels readily detectable by chemical and physical analytical techniques. Living organisms may accumulate levels of these chemicals that are greatly in excess of the ambient concentrations in their environment. The ability to accumulate substances from the environment is biologically significant, for this is how living organisms obtain those substances that are commonly designated as "essential nutrients." Non-essential chemicals also are frequently accumulated from the environment. These substances have no known biological function and may accumulate to levels that are detrimental to the organism. Trace substances may enter living organisms in several ways. Both plants and animals may accumulate these trace substances by adsorption and absorption from the external medium (air, soil, sediments, water). Animals also accumulate trace substances by ingestion.

4. The relative importance of food and direct absorption from the physical environment as pathways for entrance of trace contaminants into living organisms is the subject of considerable debate. The predominant route of entrance of a contaminant into a living organism depends on the nature of the environment itself and the relative level of exposure in the food and the external environment. Food becomes the primary source for contaminant accumulation only when direct uptake from the external environment is minimal. Another phenomenon, food-chain concentration (Odum 1971), or "biomagnification," may occur as the result of dietary intake of contaminants. At each successively higher trophic level, the concentration of a substance may increase as the result of dietary intake of food (prey) by a consumer (predator). When direct uptake from the external medium is minimal, food-chain concentration of contaminants may occur if the chemical is persistent in biological systems (Macek 1970) and the food pathway is essentially linear and highly structured, with the predominant energy flow from lower to higher trophic levels.



5. Most aquatic (freshwater and marine) ecosystems are rather weakly structured and do not have trophic levels as clearly defined as those of terrestrial systems. Energy flow in an aquatic food web is multidirectional, and a large component of the energy in aquatic systems is bound within the detritus. Aquatic systems also rarely meet the criterion that uptake from the external medium should be minimal. Contaminant levels in the water may be low, but are usually higher than levels found in the atmosphere. In comparison to terrestrial animals (terrestrial is extended to include all animals which breathe air via lungs; shorebirds and "aquatic" mammals are included as a special case of terrestrial animals living partially or wholly in water), obligate aquatic animals (i.e., gill-breathing) also have large gill areas in proportion to their body size. The solubility of oxygen in water, especially seawater, is low. Therefore, ambient oxygen available for respiration is substantially less for most aquatic animals than for their terrestrial, air-breathing counterparts. Large quantities of water must be passed over their gill surfaces to provide adequate oxygen for respiration, simultaneously increasing the contact with and uptake of both oxygen and other substances (essential and non-essential) from the surrounding medium. The body integuments of aquatic plants and animals are usually quite permeable, in direct contrast to the rather resistant integuments of terrestrial organisms. Consequently, body surfaces of many aquatic animals serve as efficient organs through which chemicals may pass into and from their tissues. The combination of intimate physical contact with the external medium, relatively permeable body surfaces, respiration via gills, and a loosely structured trophic web has led to the conclusion that trace contaminants probably do not increase nearly as much with trophic levels (i.e., biomagnify) in aquatic systems as in non-aquatic systems (Isaacs 1975). Diet is generally thought to be of minor importance as a source of most contaminants in the aquatic food web (Scura and Theilacker 1977; Macek, Petrocelli, and Sleight 1979; Narbonne 1979). Unfortunately, the majority of the studies on the accumulation of trace contaminants by aquatic animals have dealt only with uptake from the external medium. A few have studied the uptake via food alone or via food and water combined, without attempting to separate and critically evaluate the relative importance of the components of the system. Most studies addressing the uptake of contaminants with food were not designed to demonstrate cause and effect. The paucity of reliable information on the subject and the tendency to assume that

phenomena observed in non-aquatic systems occur to the same extent in aquatic systems have led to considerable confusion and controversy. Another major factor underlying this controversy is the inconsistent usage of the terms "bioaccumulation," "bioconcentration," and "biomagnification" (Macek, Petrocelli, and Sleight 1979).

#### Definitions

6. To avoid confusion, the terms "bioaccumulation," "bioconcentration," and "biomagnification" are defined below as used in this document. These definitions are cited verbatim from the paper of Brungs and Mount (1978).

Bioconcentration is usually considered to be that process by which toxic substances enter aquatic organisms, by gill or epithelial tissue, from the water. Bioaccumulation is a broader term in the sense that it usually includes not only bioconcentration but also any uptake of toxic substances through consumption of one organism. Biomagnification refers to the resultant total process including bioconcentration and bioaccumulation by which tissue concentrations of bioaccumulated toxic substances increase as this material passes up through two or more trophic levels.

The author acknowledges that the definitions of Brungs and Mount (1978) may not be entirely adequate to describe all of the processes involved in the uptake of contaminants by aquatic organisms. Other usages exist and will be treated in the text, as necessary.

#### Objectives and Scope

7. The objectives of this report are fivefold:

- a. Review the literature on biomagnification of contaminants within aquatic ecosystems;
- b. Assess the relative importance of food as a source of contaminants in aquatic food webs;
- c. Pinpoint those contaminants which may substantially biomagnify within aquatic food webs;
- d. Indicate the gaps in existing knowledge; and
- e. Provide recommendations for further research on biomagnification of contaminants in aquatic systems, based upon the findings of this report.

This report constitutes a selected review of the literature on the trophic uptake and biomagnification of contaminants through successive trophic levels involving gill-breathing aquatic animals. It is beyond the scope of this report to survey the voluminous literature treating toxicology or bioconcentration of contaminants in aquatic systems. Other literature shall be included if it provides strong circumstantial evidence that either supports or discounts biomagnification as a significant problem in the aquatic system. The potential for biomagnification in aquatic food webs also is presumed to be independent of the route of entry of contaminants into the primary consumer. The consumption of aquatic organisms by humans and public health implications will not be discussed. Contaminant concentrations in organisms are reported on a dry weight basis unless otherwise stated.

## PART II: REVIEW OF THE LITERATURE

### Heavy Metals

#### Cadmium

8. Cadmium in freshwater organisms. Laboratory studies suggest that cadmium (Cd) does not biomagnify in aquatic systems, even though food may be an important source of Cd for aquatic fauna. Hatakeyama and Yasuno (1981) fed *Chlorella* sp. (phytoplankton) containing various levels of Cd to the cladoceran, *Moina macrocopa*. After 7 days of feeding on *Chlorella* containing 240 ppm Cd, the Cd concentration in *Moina* was about 85 ppm. Bioconcentration from water containing 20 ppb was about 123 ppm in *Moina* after only 2 days. The accumulation of Cd by *Moina* was essentially linear, whether the source of Cd was food or the water. Hatakeyama and Yasuno (1982) extended their earlier study to include a Cladoceran (*Moina*)-guppy (*Poecilia reticulata*) food chain and compared the uptake of Cd from food and water. After 6 days, the bioconcentration of Cd by the guppy from static water containing 45 ppb Cd was about four times that for accumulation after 30 days from food (*Moina*) containing 171 ppm Cd. In both of these studies, the rate of uptake from water was significantly greater than from food. No data were included to show Cd depuration rates following exposure to Cd either via food or water. In a wastewater-based, artificial food chain, Tarifeño-Silva et al. (1982) reported that *Daphnia* spp. accumulated only 2 to 5 ppm Cd when fed *Scenedesmus* spp. containing 60 ppm Cd (Table 1).

9. Field studies also generally indicate no increase of Cd with increasing trophic level. Mathis and Cummings (1973) reported Cd levels in the Illinois River decreasing in the order sediment > annelids > clams > fishes > water. Predatory fishes had similar or lower Cd levels than omnivorous fishes. The Cd levels in fishes were 1 to 2 orders of magnitude lower than those in annelids and clams. Thomann et al. (1974) developed a five-component food-chain model of Cd for Lake Erie that included water, phytoplankton, zooplankton, fish, and birds. The model indicated that Cd concentrations increased with trophic level and was supported by rather limited data in which Cd concentrations in organisms were expressed in terms of micrograms per milligram of organic carbon (OC). Summary data for Cd in western Lake Erie indicated Cd concentrations in goldfish, yellow perch, white bass, walleye, and

spottail shiner averaged 1,400 ( $\pm 1100$ ), 500 ( $\pm 60$ ), 200, 200, and 100 ( $\pm 30$ )  $\mu\text{g Cd/g tissue}$ . The top predators (bass and walleye) in this system had lower mean Cd levels than goldfish, which are basically omnivorous and may be prey for the carnivores. This suggests that biomagnification of Cd does not occur. The model developed by Thomann et al. (1974) in this paper does not consider bioconcentration from the water and appears to be based largely on theory without adequate data to support the numerous assumptions included in the model.

10. Enk and Mathis (1977) reported Cd levels in fish invertebrates, water, and sediments from Jubilee Creek in Illinois. Table 2 shows their data presented with the food habits of the species analyzed and indicates no correlation between Cd content in tissues and trophic level. Fish contained Cd levels similar to those in the sediments. The highest levels, 1.54 and 1.19 ppm (wet weight), respectively, for the damselfly, *Agrion* (predator), and the mayfly, *Isonychia* (detritivore), probably were not significantly different than those observed for the herbivorous caddisflies, *Hydropsyche* and *Cheumatopsyche*. Since the insects all contained significantly greater Cd concentrations than did the fish, and predatory fish had similar Cd levels as other fish, one can conclude that food contributed relatively little to the Cd body burden of the fish.

11. Anderson (1977) studied the bioaccumulation of Cd in 21 genera of freshwater invertebrates collected in the Fox River, Illinois. Largely predaceous insects (Odonata, Hemiptera, and Coleoptera) contained  $\leq 0.5$  ppm Cd, whereas filter-feeding, detritivorous, and omnivorous benthic invertebrates (insects, molluscs, and crustaceans) generally had substantially greater Cd levels (Table 3). One leech, *Erpobdella*, had  $3.80 \pm 0.30$  ppm Cd, while another, *Placobdella*, had  $\leq 0.5$  ppm Cd.

12. Cherry and Guthrie (1977) measured the concentrations of Cd in water, sediments, and organisms before and after dredging an ash basin in South Carolina. The order of increasing mean Cd concentrations before dredging was water < vertebrates < plants < sediment < invertebrates. Following dredging, the concentrations of Cd decreased in all biota, approximately doubled in the sediments, and remained approximately the same in the water. Valid conclusions are difficult to make from these data, as the biotic components did not constitute a logical food chain.

13. Kneip and Hazen (1979) investigated the deposition and mobilization of Cd in a marsh-cove ecosystem in connection with dredging of Cd-contaminated

sediments in the vicinity of a nickel-cadmium battery plant. The general order of increasing Cd concentration was fish < amphipods < macrophytes < plankton < sediments. From comparisons of organ-burden distributions for field-captured *Fundulus diaphanus* and "implanted" goldfish (*Carassius auratus*) with laboratory data, they concluded that food must be a major route of Cd exposure for fish under field conditions. Although they reported the concentrations of Cd in the food and water were 24 µg/g and 250 µg/l, respectively, they neither reported concentrations in tissues nor gave information on the methodology used to compare the ingestion of Cd in food with bioconcentration from water.

14. May and McKinney (1981) reported Cd in fish varying from <0.01 to 1.04 ppm. The higher value was for carp (benthic omnivore) collected in the Des Moines River at Keosauqua, Iowa; walleye (predator) caught at this location contained <0.05 ppm Cd, however.

15. Cadmium in marine organisms. Laboratory studies by Boothe and Knauer (1972) suggested that fecal material could have an important role in the "biological amplification" of trace metals in the marine environment. Feces from crabs (*Pugettia producta*) fed exclusively on the Cd-contaminated alga, *Macrocystis pyrifera*, contained only 13 percent of the Cd ingested, however. The fate of the remaining 87 percent of the Cd ingested is uncertain, suggesting either retention in the crab or excretion through a means other than in feces. Pentreath (1977) examined the uptake of Cd from <sup>115</sup>Cd-labelled seawater and food (*Nereis*) by plaice (*Pleuronectes platessa*) and thornback ray (*Raja clavata*). Uptake of <sup>115</sup>Cd from water containing 2 g Cd/l was very slow and linear with time in both species. Plaice and rays fed a single <sup>115</sup>Cd-labelled *Nereis* at day 0 and one unlabelled *Nereis* daily beginning at day 6 retained about 4 and 17 percent (indicated by Pentreath; his data showed about 30 percent) of the ingested <sup>115</sup>Cd after 41 days. Pentreath indicated that "both species retained cadmium from food and accumulated it in the liver" and, further, that uptake from seawater was less important for the ray than for plaice. Pentreath did not specify the concentrations of either <sup>115</sup>Cd or Cd in the labelled *Nereis*, but only indicated that the Cd level in unlabelled *Nereis* was about 0.1 µg Cd/g wet weight. Consequently, the author of the present review could not follow Pentreath's reasoning and was unable to find any clear comparison between food and water as routes of entry of Cd into either species.

16. Analyses of field-collected organisms generally have shown no

biomagnification of Cd in marine food webs. Preston et al. (1972), however, found that concentrations of Cd in a limpet (*Patella* sp.) exceeded those in algae (*Porphyra* spp. and *Fucus* spp.). The concentrations of Cd varied from 2.8 to 23 ppm in *Porphyra*, and 0.5 to 3.0 ppm in *Fucus* (Table 4). This

implied possible trophic movement and potential biomagnification of Cd, provided the limpets actually were feeding on the algae (not indicated by the authors). Leatherland et al. (1973) reported Cd concentrations in fish and various invertebrates ranging from 0.05 µg/g in muscle of the skipper (*Scom bresox saurus*) to 13.0 µg/g in two decapod crustaceans (*Systellaspis debilis* and *Oplophorus* sp.). The lowest trophic levels (according to the authors) were represented by an omnivorous euphausiid (*Meganyctiphanes norvegica*) and an herbivorous tunicate (*Pyrosoma* sp.) containing 0.25 and 0.44 µg Cd/g, respectively (Table 5). There was no discernible relationship between Cd concentration in tissues and trophic level.

17. In a turtle grass (*Thalassia testudinum*) community in Card Sound, Florida, Gilio and Segar (1976) found no indication of increased Cd concentration with increasing trophic levels. The ranges of mean Cd concentrations were 0.11 (±0.012) to 0.20 (±0.047) in macrophytes, 0.20 in phytoplankton and in epiphytes on *Thalassia* leaves, 0.19 (±0.08) in combined detritivores and carnivores, and 0.44 (±0.18) in sponges (Table 6). The higher levels of Cd in sponges (filter-feeders) suggest that food (phytoplankton) may be the source of Cd in sponges, but differences do not appear to be statistically significant. A compartment model for trace elements showed that total Cd load increased from biota to water to sediments. A similar study by Talbot and Cheg-widden (1982) measured levels of Cd in seagrasses (*Posidonia* sp.) and their epiphytes, sea lettuce (*Ulva lactuca*), mussels (*Mytilus edulis*), oysters (*Ostrea angasi*), polychaete worms (*Chaetopterus variopedatus*), and crabs (*Portunus pelagicus*). A comparison of the data collected at the same stations reveals that the concentrations of Cd in mussels and polychaetes are similar, whereas those in crab (hepatopancreas) are about an order of magnitude greater (Table 7). On the basis of Cd in flesh (muscle), the crabs also had similar Cd levels as did mussels and polychaetes, however. As mussels often are part of the diet of crabs, mussels may have contributed to the Cd content of the crabs. The Cd concentrations in the seagrasses and their epiphytes and in sea lettuce were generally less than or equal to those in the invertebrates. Amiard et al. (1980) reported that there was no evidence for biomagnification

of Cd within an estuarine food web. Cadmium concentrations were greater in crustaceans than in whole fish (Table 8). Gut contents of fish contained significantly greater Cd levels than did whole fish (gut contents excluded), and Cd was more concentrated in the intestines than in the stomach. The authors stated that this was the result of "faeces enrichment by unassimilated metals." On the basis of fresh weights and dry: fresh weight ratios given by the authors, the reviewer calculated the percent total body burden of Cd contained within the gut for each organism for which gut concentrations were given. These data indicated that 90 percent of the total body burden of Cd was in the tissues and only 10 percent was in gut contents. The percent of total body burden of Cd contained in the stomach was approximately equal to that in the intestine, however, suggesting that little Cd was absorbed in the stomach.

18. Others have attempted to relate Cd levels to feeding habit. Bryan and Hummerstone (1977) showed no clear relationship between feeding habit and Cd concentration in several marine gastropods, pelecypods, and one polychaete. The concentrations of Cd in the deposit-feeding polychaete (*Nereis versicolor*), deposit-feeding pelecypods (*Scrobicularia plana* and *Macoma balthica*), filter-feeding pelecypods (*Cerastoderma edule* and *Mytilus edulis*), and an herbivorous gastropod (*Littorina littorea*) were not greatly different than that in the brown alga, *Fucus* sp., but all were substantially elevated in contrast to the level of Cd in the sediments (Table 9). Cadmium levels in the limpet, *Patella vulgata* (herbivore), and the dogwhelk, *Nucella lapillus* (predator), were substantially elevated over those in the other species and averaged 8.6 and 12.8 ppm, respectively (Table 10). Cutshall, Naidu, and Percy (1977) suggested that Cd in Pacific hake (*Merluccius productus*) reflected its diet, primarily euphausiids, which contained "relatively high contents of" Cd. Cadmium levels in hake muscle and whole fish averaged 0.03 and 0.12 ppm, respectively, and, in euphausiids, 0.23 ppm. These data suggest that the trophic transfer of Cd from euphausiids to hake was minimal. The data for Cd in hake and euphausiids were collected separately, however. Consequently, the authors' suggestion of a diet-related source of Cd for Pacific hake must be accepted with reservation. Cadmium analyses of pink shrimp (*Pandalus jordani*) and several other fish species showed no pattern with respect to feeding habit and suggested that biological magnification of Cd was unlikely, as levels in the euphausiids consistently exceeded those in the fish.

19. Recent studies have used the cesium:potassium (Cs:K) ratio to rank



marine organisms according to trophic levels. Young and Mearns (1979) measured Cd levels in the biota of the Salton Sea and two marine food webs along the coast of California. The Cd concentrations did not increase with trophic level in any of these food webs (Table 10). Similarly, Schafer et al. (1982) examined the Cd concentrations in organisms of different trophic levels in a coastal pelagic and an epibenthic food web and found no evidence for biomagnification of Cd in either instance (Table 11). In the epibenthic food web, Cd levels were greatest in the sediments, decreased in mysids and decapods, and further decreased in several species of fish and in ridgeback prawn. In the pelagic food web, zooplankton had the highest Cd levels, followed by squid and anchovy (0.172). The lowest value observed was 0.004 ppm in the white shark, the top predator in this food web.

#### Tin

20. Very little information is available on tin (Sn) in aquatic food webs. A laboratory study (Tarifeño-Silva et al. 1982) indicated that the concentration of Sn decreased in the order algae > microcrustaceans > fish in a wastewater-based artificial food chain (Table 1). The algae *Scenedesmus* spp. and two microcrustaceans, *Daphnia magna* and *D. pulex*, averaged 70, 30, and 0.6 ppm, respectively. Golden shiners (*Notemigonus chrysoleucas*) and fathead minnows (*Pimephales promelas*) maintained for 7 weeks on mixed diets of the two species of daphnids averaged 3.91 and 3.63 ppm Sn, respectively.

21. Jenkins (1976) briefly lists data (compiled from numerous other studies) on the maximum concentrations of metals reported in organisms and then computes an enrichment factor. Maximum concentrations of Sn reported were 3.8, 2.3, 32, 15, and 5.4 ppm (wet weight), respectively, for algae, higher plants, invertebrates including crustaceans, molluscs, and fish. The relative enrichment factors computed for Sn in marine organisms were 0.43, 3.5, and 0.5, respectively, for algae, invertebrates (excluding molluscs), and fish. Phillips et al. (1982) reported that Sn concentrations were <5.0 ppm in fish, molluscs, and crustaceans obtained from retail markets in Hong Kong. Maximum values of 13.8 and 17.4 ppm Sn, respectively, were reported for samples of lobsters and crabs in the market at Lau Fau Shan on 20 January 1978. These data are insufficient to make valid conclusions about the potential for biomagnification of Sn in aquatic food webs, however.

#### Selenium

22. Relatively little information is available on the uptake of

selenium (Se) from food in aquatic food webs. The contribution of gut contents to the total body burden of Se in the detritus-feeding larvae of a crane-fly (*Tipula* sp.) was examined by Elwood, Hildebrand, and Beauchamp (1976). The Se concentrations in whole larvae, larvae following evacuation of gut contents, and feces were 1.02, 0.95, and 0.99 ppm, respectively (Table 12). Selenium in the gut contents represented 23 percent of the total body burden. The concentration of Se in leaf detritus was 0.49 ppm. A trophic transfer factor  $\frac{\text{Se in larvae with gut evacuated}}{\text{Se in leaf detritus}}$  calculated by Elwood, Hildebrand, and

Beauchamp (1976) was 1.9, suggesting that food-chain enrichment (biomagnification) of Se might occur. A valid conclusion was difficult to attain without information for higher trophic levels, but food did appear to be an important source of Se in *Tipula* larvae. Cherry and Guthrie (1977) reported that Se was biomagnified by organisms in an ash basin. The order of mean increasing concentrations of Se was water < plants < invertebrates < sediments < vertebrates (Table 13). Each group of biota was mixed in terms of types of organisms included: plants included varied from algae to trees; invertebrates varied from midges to crayfish; vertebrates included were mosquitofish (*Gambusia affinis*) and frog tadpoles (*Rana* sp.). The authors' conclusion of biomagnification of Se was greatly compromised by the inclusion of organisms that did not constitute a logical food chain and, consequently, must be accepted with reservation. In a compendium of data collected for the National Pesticide Monitoring Program during 1976-1977, May and McKinney (1981) found no pattern to suggest any relationship between Se concentration in fish and presumed trophic level. Concentrations of Se in fish varied widely with location, and only rarely were more than two species analyzed from the same area. The Se concentrations reported ranged from <0.05 ppm in Klamath sucker to 2.87 ppm in goldeneye. Using Cs:K ratios to estimate trophic level, Schafer et al. (1982) found no relationships between Se concentrations in biota and position in the food web (Table 11). In a tropical pelagic food web, Se concentrations ranged from 0.82 ppm in zooplankton to 0.96 ppm in the silky shark, the top predator examined. In a coastal pelagic food web, Se levels were 0.45 and 0.48 ppm, respectively, in the white shark (top predator) and zooplankton. The highest levels reported were 1.15, 2.11, and 2.16 ppm, respectively, in jack mackerel, bonita, and squid, all of which fell in the intermediate trophic levels.

23. Although the data available suggest that Se may be obtained from

food by aquatic animals, it is doubtful that biomagnification occurs in aquatic food webs.

#### Silver

24. Very little information is available related to the potential for the biomagnification of silver (Ag) in aquatic food webs. Preston et al. (1972) compared the concentrations of Ag in seawater, two species of algae (*Fucus vesiculosus* and *Porphyra umbilicalis*), and in herbivorous limpets (*Patella vulgata*). The average Ag concentration in limpets was an order of magnitude greater than those in the algae (Table 4), which suggests that Ag could possibly biomagnify. The authors indicated that the limpets were collected from rocks, however, suggesting that the food source was more likely epilithic organisms than the macroalgae. Data reported by Bryan and Hummerstone (1977) did not show any biomagnification of Ag in the Looe Estuary (Table 9). The predatory dogwhelk (*Nucella lapillus*) contained an average of 2.7 ppm Ag, in comparison with the herbivores *Patella vulgata* (limpet) and *Littorina littorea* (periwinkle), which averaged 3.0 and 19.6 ppm Ag, respectively. Deposit-feeders, particularly clams, contained substantially higher levels of Ag than the dogwhelk, limpet, or filter-feeders. Rice, Tenore, and Windom (1981) reported that Ag concentrations in deposit-feeding polychaetes (*Capitella capitella*) decreased with increasing detrital rations. Detrital rations were expressed as g nitrogen (N)  $\text{m}^{-2} \text{day}^{-1}$ . Initial Ag concentrations in the worms, *Ascomyllum detritus*, and *Gracilaria detritus* averaged  $0.78 \pm 0.04$ ,  $0.28 \pm 0.03$ , and  $0.19 \pm 0.02$  ppm, respectively. At detrital rations of approximately 0.05 and 0.10 g N  $\text{m}^{-2} \text{day}^{-1}$ , respectively, worms fed either *Gracilaria detritus* or *Ascomyllum detritus* contained approximately 2.1 and 0.5 ppm Ag. The authors stated that the decrease in the uptake of Ag by the worms with increasing detrital ration was due to "increased reworking of the total benthic trace metal pool (food, sediments, water) by larger populations of *C. capitata*." The meaning of their statement is unclear, however. Presumably, they meant that the apparently greater uptake at the lower detrital rations was the result of more efficient extraction of Ag from the detritus, as the worms would have had to reprocess the detritus more extensively to obtain adequate food (initial worm biomass:detrital ration ratio was greater at the lower detrital ration). Then, at higher detrital rations (i.e., lower initial worm biomass:detrital ratio), the worms apparently had more food and did not have to reprocess the detritus extensively, resulting in depuration of

Ag instead of uptake. The relative uptake of Ag from food by detritus-feeding polychaetes, therefore, appears to be dependent upon the amount of reprocessing of the detritus.

25. Using Cs:K ratios to rank organisms within trophic levels, Young and Mearns (1979) found no increase in Ag concentration with increased trophic levels (Table 10). In the Salton Sea and Newport Bay food webs, Ag levels were relatively stable (0.002 to 0.003 ppm) across all trophic levels. The Ag levels in the Palo Verdes food web varied from <0.003 ppm in scallops to 0.095 ppm in the yellow crab. The highest and lowest trophic levels were represented by bocaccio and abalone, respectively, and contained 0.008 and 0.028 ppm Ag. Similar studies by Schafer et al. (1982) demonstrated no relationship between trophic level and Ag concentration in California coastal pelagic or epibenthic food webs (Table 11). The Ag concentrations in the epibenthic food web were 0.167 ppm in mysids and decapods (lowest trophic level) and <0.001 to <0.002 at all other trophic levels. In the pelagic food web, the highest levels of Ag were 0.12 ppm in squid (an intermediate trophic level) and 0.004 ppm in the white shark (top predator). These data suggest that biomagnification of Ag over several trophic levels does not occur in aquatic food webs.

#### Nickel

26. Laboratory studies suggest that some uptake of nickel (Ni) from food may occur, but biomagnification of Ni probably does not occur. Rice, Tenore, and Windom (1981) reported that Ni concentrations in the deposit-feeding polychaete worm, *Capitella capitata*, decreased with increasing detrital ration of the seaweeds, *Gracilaria foliifera* and *Ascophyllum nodosum*. Initial Ni concentrations in worms, *Gracilaria detritus*, and *Ascophyllum detritus*, were 7.3, 2.4, and 13.1 ppm, respectively. Worms fed for 90 days on *Ascophyllum* or *Gracilaria detritus* rations of approximately 0.05 and 0.10 g N m<sup>-2</sup> day<sup>-1</sup> contained 14 and 8 ppm Ni, respectively. Increased Ni uptake at lower detrital rations probably was due to reprocessing of the detritus by the worms, as discussed previously in the Ag section. Data for algae (*Scenedesmus* sp.), cladoceran crustaceans (*Daphnia magna* and *D. pulex*), and fish (*Notemigonus crysoleucas* and *Pimephales promelas*) raised in wastewater-based artificial food chains suggest that Ni is probably not biomagnified in freshwater food webs (Tarifeño-Silva et al. 1982). The Ni concentrations in algae, *D. magna*, *D. pulex*, and in muscle tissue of *N. chryssoleucas* and *P. promelas*

averaged 15, 3, 1.7, 4.45, and 2.71 ppm, respectively (Table 1). Total body burdens of Ni in the fishes were not given.

27. There is some evidence for the uptake of Ni under field conditions. Preston et al. (1972) indicated that the Ni concentrations in the limpet, *Patella vulgata*, were generally higher than those in macroalgae, *Porphyra umbilicalis* and *Fucus* sp. (Table 4). These data suggest that Ni could be biomagnified in the limpet, providing the limpets actually fed on *Porphyra*. As the authors stated that the limpets were collected from rocks, one cannot conclude that Ni was biomagnified. Kay and Rojanavipart (1976) reported that Ni concentrations in the periwinkle (*Littorina littorea*) were elevated in comparison with those in the stems and leaves of common cord-grass (*Spartina anglica*), common saltmarsh-grass (*Puccinellia maritima*), and a dwarf shrub (*Halimione portulacoides*). Periwinkles commonly feed on the epiphytes and occasionally on saltmarsh vegetation, such as these species. The Ni concentrations in periwinkles were similar to those in macroalgae (*Fucus* sp.) and detritus and lower than those in the sediments. The relative contributions of the direct uptake of Ni from the environment (e.g., through the foot from the sediments or detritus) and uptake via food could not be determined.

28. Other field studies indicate that there is little movement of Ni through increasing trophic levels and that biomagnification of Ni probably does not occur within aquatic food webs. Mathis and Cummings (1973) reported that the order of increasing Ni content in the Illinois River was water < carnivorous fishes < omnivorous fishes < clams < tubificid worms < sediment (Table 14). Nickel appears to remain closely associated with the sediments and less so with the water column. Jenkins (1976) compiled from the literature the maximum concentrations of Ni in biota and calculated "maximum enrichment factors" for Ni in marine organisms. The enrichment factors (relative to an average Ni concentration of 2 ppb in water) were 3,000, 6,500, 2,500, 7,900, and 300, respectively, for algae, zooplankton, invertebrates (except molluscs), molluscs, and fish. Bryan and Hummerstone (1977) found that the predatory dogwhelk (*Nucella lapillus*) contained similar Ni concentrations (Table 9) as those in herbivorous gastropods (*Littorina littorea* and *Patella vulgata*), a filter-feeding mussel (*Mytilus edulis*), and a deposit-feeding polychaete worm (*Nereis diversicolor*). The Ni concentrations in brown algae (*Fucus* sp.) were higher than those in the herbivorous gastropods and similar to those in deposit-feeding clams (*Scrobicularia plana* and *Macoma balthica*). Highest Ni

levels were found in a filter-feeding cockle (*Cerastoderma edule*), which contained Ni concentrations similar to those of the sediments. Thus, there was no apparent relationship between Ni concentrations and feeding habit and no trend toward biomagnification of Ni in this food web.

29. Using Cs:K ratios to establish trophic levels, Young and Mearns (1979) demonstrated no trend toward biomagnification of Ni in any of three saltwater food webs (Table 10). In the Salton Sea and Newport Bay food webs, Ni concentrations averaged <0.03 to <0.04 ppm. In the Palo Verdes food web, the highest Ni level (0.68 ppm) was found in abalone, the lowest trophic level in this food web. Using similar techniques, Schafer et al. (1982) found that the concentrations of Ni in an epibenthic and a coastal pelagic food web (Table 11) were lower in the top carnivores than in lowest trophic levels. In the pelagic food web, white sharks and zooplankton contained <0.019 and 0.294 ppm, respectively. In an epibenthic food web at Palos Verdes, Ni concentrations in scorpionfish and combined samples of mysids and decapods were <0.030 and 1.07 ppm, respectively. The Ni levels in intermediate trophic levels varied somewhat, with no pattern to suggest any relationships to feeding habits.

#### Arsenic

30. Arsenic in freshwater organisms. Information on arsenic (As) in freshwater food chains is rather limited, but the available data suggest that As does not biomagnify. Cherry and Guthrie (1977) reported the concentrations of As in water, sediments, and biota in a coal ash basin before and after dredging (Table 13). Prior to dredging, the As concentrations decreased with increasing trophic level. Following dredging, As levels in plants remained approximately constant, but those in invertebrates and vertebrates increased about 30-fold and 8-fold, respectively. The importance of trophic intake and potential for biomagnification of As could not be determined, as the organisms included did not form a logical food chain. May and McKinney (1981) reported As levels in freshwater fish ranging from 0.05 to 2.92 ppm (wet weight) and averaging 0.27 ppm. Arsenic levels varied widely with location, with no discernible relationship between tissue concentrations and presumed trophic levels. At stations where species sampled represented several obvious trophic levels, As levels were generally lower at the upper trophic level, suggesting that As does not biomagnify in the food chain.

31. Arsenic in marine organisms. Laboratory studies indicate that As

may be absorbed from contaminated foods. Booth and Knauer (1972) reported that fecal concentrations of arsenic in crabs feeding on algae were elevated in comparison to those in the algae. The theoretical retention of As in fecal pellets was about 32 percent (Table 15), suggesting that As might be readily assimilated into the crabs. The concentration of As in the crabs and the total consumption of algae by the crabs were not indicated, however. Consequently, no valid conclusions can be made from this study either on bioaccumulation or biomagnification of As. Klumpp (1980) studied As accumulation from food and water and the subsequent depuration of As by periwinkles (*Littorina littoralis*) and dogwhelks (*Nucella lapillus*) using  $^{74}\text{As}$ . Periwinkles exposed to  $^{74}\text{As}$  via food ( $^{74}\text{As}$ -labelled *Fucus*) contained 20 percent of the  $^{74}\text{As}$  activity in the foot-buccal mass, 80 percent in digestive glands and gonads, and none of the  $^{74}\text{As}$  in the shell and operculum. Exposure of periwinkles to  $^{74}\text{As}$  via water resulted in about 1 percent of the  $^{74}\text{As}$  activity in the operculum, 9 percent in the shell, 4 percent in the foot-buccal mass, and the remainder in the digestive gland and gonads. The specific  $^{74}\text{As}$  activity in periwinkles was about five times that in their food (labelled *Fucus spiralis*) after 21 days of feeding. During a 4-day depuration period, the loss of  $^{74}\text{As}$  activity in water-labelled and food-labelled periwinkles was dependent on food consumption during the depuration period. Periwinkles fed unlabelled food (*Fucus*) during depuration lost about 45 to 50 percent of their total  $^{74}\text{As}$  activity in contrast to about 15 percent loss in those which were unfed during depuration. In the predatory dogwhelk (*Nucella lapillus*), the contribution of  $^{74}\text{As}$  uptake from water to the total As pool in the whelks was small, in comparison with uptake from labelled food (i.e., periwinkle). The authors calculated that about 0.8 percent of the total As pool of the whelks would come from direct absorption from water containing 3 ppb As. No account was made for possible direct As absorption through the foot of either species, however. The data do suggest strongly that contaminated food may be an important route for As entry into the lower trophic levels of marine food webs.

32. Field studies generally have indicated no relationship between trophic level and As concentration in organisms, however. Leatherland et al. (1973) reported that As levels in pelagic marine organisms varied greatly and showed no particular pattern with respect to trophic levels. Euphausiids and an herbivorous tunicate, representing the lowest trophic level, and a shark, the highest trophic level, contained 42, 1.5, and 1.2 to 1.3 ppm As,

respectively (Table 5). Organisms designated by the authors as belonging to intermediate trophic levels had As content ranging from 2.5 to 30 ppm. Jenkins (1976) averaged data from the literature for maximum As concentrations in aquatic organisms and calculated enrichment factors for marine organisms. The maximum As levels increased in the order: fish < higher plants < algae < invertebrates. With respect to a mean As concentration of 2 ppb in the ocean, the "maximum enrichment factors" for marine algae, fish, invertebrates (except molluscs), and molluscs were 75,000, 10,000, 36,000, and 40,000, respectively. Greig et al. (1977) examined the occurrence of trace metals in marine organisms at dredged material ocean disposal sites (Table 16). Arsenic concentrations in crabs and in flounder at the Delaware site were 1.9 and 1.8 ppm, respectively. The As content in different organisms varied with location. The lowest levels reported were 1.3 ppm in muscle of clams at the control site (Chincoteague Inlet) and 1.4 ppm in flesh of flounder at the New York Bight dumpsite. The maximum As level reported was 9.0 ppm in the muscle of channeled whelks (predatory) at Long Island Sound. The data in this study were insufficient to make any valid conclusions on As levels in the biota near dumping sites.

33. Edmonds and Francesconi (1981) measured As levels in marine organisms at Waterman off the coast of western Australia. The As concentrations in plants, fish (*Sillago bassensis*), fish gut contents (largely polychaete worms), and polychaete worms ranged from <0.1-15.9, 3.2-14.5, 1.3-31.3, and 7.1-23.0 ppm, respectively (Table 17). The authors imply a possible trophic relationship between As in the plants and As in the fish: plant → detritus → worms → fish. The fish, however, fed primarily on polychaetes, some of which (according to the authors) were not detrital feeders. The data were inadequate to allow any valid conclusions as to the source of As for the fish. Using the Cs:K ratios to determine trophic levels, Schafer et al. (1982) found that As levels were elevated in sharks at the upper trophic levels (Table 11). In a coastal pelagic food web, the As levels in white sharks and mako sharks were 3.18 and 3.53 ppm, respectively. The two lowest trophic levels, anchovy and zooplankton, contained 1.66 and 1.32 ppm As, respectively. Arsenic in intermediate trophic levels varied from 0.19 in the blue whale to 0.81 in sardines. A similar pattern was observed in a tropical pelagic food web, where As levels in silky sharks were 4.1 ppm. The authors indicated that, when the shark data were omitted, there was no obvious relationships between As



concentrations and trophic level in either food web. These data suggest that food may be an important route of As uptake in the large, predatory sharks. Otherwise, there were no clear indications of the biomagnification of As in marine food webs.

#### Chromium

34. Chromium in freshwater organisms. Laboratory data indicate that very little chromium (Cr) is absorbed from food by freshwater animals. Patrick and Loutit (1976) reported that tubificid worms could accumulate Cr by the ingestion of contaminated bacteria (Table 18). Worms fed for 7 days on bacteria containing 109, 983, and 2,850 ppm Cr and allowed to depurate 24 hr to evacuate gut contents accumulated only 3.9, 14.1, and 29.9 ppm Cr, respectively. The concentrations of Cr in the worms represented only 1.1 to 3.5 percent of the levels in the bacteria, and only those worms fed on bacteria containing the highest Cr levels contained more Cr than the original field-collected worms. Elwood, Hildebrand, and Beauchamp (1976) demonstrated that gut contents represent a substantial portion of the body burden of Cr in detritus-feeding *Tipula* sp. larvae (Table 12). The concentrations of Cr in the larvae were about one third of that in leaf detritus. The concentration of Cr in the feces was similar to that of the leaf detritus, however. Chromium concentrations in the larvae also did not change significantly following gut evacuation, suggesting that relatively little Cr was absorbed from the food. Magnuson et al. (1980) studied the uptake of Cr by crayfish fed leaf discs containing approximately 450 ppm Cr. After 8 weeks of feeding, the mean Cr concentration in the crayfish was about 40 ppb. The authors reported that the crayfish retained only 1.72 percent of the total Cr ingested during the 8-week feeding period. Tarifeño-Silva et al. (1982) fed cladoceran crustaceans (*Daphnia* sp.) on the unicellular green alga, *Scenedesmus* spp., in an Fe-enriched wastewater effluent. The Cr levels in water, algae, *D. magna*, and *D. pulex*, were 0.01, 13, 3, and 8 ppm, respectively. Although the relative importance of Cr uptake from the water and food was difficult to discern, Cr obviously did not biomagnify in this single-step food chain.

35. Field studies seem to confirm that trophic transfer of Cr does not occur in freshwater ecosystems. Mathis and Cummings (1973) reported the levels of Cr in water, sediments, and biota from the Illinois River. The Cr concentrations decreased in the order sediments > tubificid worms > clams > fishes (Table 14). Similar Cr levels were found in both predatory and

omnivorous fishes. Cherry and Guthrie (1977) demonstrated a slight increase of Cr concentrations from plants to aquatic invertebrates both before and after dredging a coal ash basin (Table 13). Before dredging, average Cr levels in vertebrates were only a third the levels in the invertebrates. Following dredging, however, Cr doubled in vertebrates and decreased in both plants and invertebrates. After dredging, Cr levels in the vertebrates were about twice those in either plants or invertebrates, very likely as the result of increased exposure via suspended sediments. Biomagnification of Cr probably did not occur. No valid conclusion can be made as the organisms did not constitute a logical food chain.

36. Chromium in marine organisms. Laboratory work with marine invertebrates indicates some possible uptake of Cr from foods. Preston (1971) used radioactive  $^{51}\text{Cr}$  to examine the relative roles of food and direct absorption in the uptake of Cr by oysters (*Crassostrea virginica*). Oysters were exposed in a closed system without renewal of  $^{51}\text{Cr}$  sources to 50 microcuries  $^{51}\text{Cr}$  per liter ( $\mu\text{C}/\ell$ ) in artificial seawater or in a suspension of the green microalga, *Chlamydomonas* sp., previously labelled for 5 days with  $^{51}\text{Cr}$  to provide 50  $\mu\text{C}$   $^{51}\text{Cr}/\ell$  associated with algal cells. After 110 hr of contact, the accumulations of  $^{51}\text{Cr}$  by oysters were about 1,100 DPM (disintegrations per minute) and 300 DPM, respectively, for uptake from water and food. The distribution of  $^{51}\text{Cr}$  in different tissues was similar, regardless of route of entry. The authors acknowledged that leakage from algal cells probably provided a portion of the  $^{51}\text{Cr}$  absorbed from the food, but suggested that food might be a more important source of  $^{51}\text{Cr}$  in nature, however, as "radioactivity is likely to be greater in the food supply than in the water since most organisms tend to concentrate radionuclides." Boothe and Knauer (1972) suggested that food may be important in the transfer of Cr within marine food webs. Crabs (*Pugettia producta*) fed exclusively on the brown alga, *Macrocystis pyrifera*, contained similar Cr concentrations in their feces as in their food source (Table 15). Of the calculated theoretical total weight of metal ingested, only 11 percent remained in the feces. This implies that about 89 percent had been retained within the crabs, but does not account for other potential routes for loss of Cr. The Cr concentrations in the crabs were not reported. Although both of these laboratory studies suggest uptake of Cr from food, neither provides evidence that biomagnification of Cr occurs.

37. Field studies, however, suggest that very little Cr is passed on

through food. Jenkins (1976) calculated enrichment factors (as compared to seawater) for Cr in different groups of marine organisms. These factors were 17, 25, 25, 320, and 2.5, respectively, for algae, zooplankton, invertebrates (except molluscs), molluscs, and fish. These data imply that some Cr enrichment occurs at the lower trophic levels, but little transfer of Cr to the upper trophic levels will occur. A study of Cr in sediments and biota of the Looe Estuary suggests that some Cr may be taken in via food, but biomagnification probably does not occur within this marine food web (Bryan and Hummerstone 1977). The mean concentrations of Cr were highest in the sediments, followed by deposit-feeding clams, algae, the predatory dogwhelk, filter-feeders, herbivores, and a deposit-feeding polychaete worm (Table 9). No data were available for Cr in biota in higher trophic levels. Greig et al. (1977) reported Cr levels in biota and sediments at several ocean dumping sites for dredged material (Table 16). Chromium was more concentrated in the gills of crabs than in other organs. Within a single site, concentrations of Cr in the flesh of different organisms were quite similar, suggesting that Cr does not biomagnify within the food web. Heavy metals were examined in fishes from the Chao Phraya River Estuary in Thailand (Polprasert 1982). The average Cr concentrations reported for trophic levels III and IV were 9.55 and 12.27, respectively. Polprasert contended that this represented "possible biomagnification" of Cr. The ranges of Cr concentrations were 0.92 to 48.64 ppm at level III and 1.25 to 40.68 at level IV. Many species were included in the study, with no reference either to feeding habits or to how trophic levels were assigned. The data as presented by the authors appear inadequate to make any statements concerning the possible occurrence of biomagnification of Cr. Chromium was reported in trace quantities in fish and shellfish from Hong Kong (Phillips et al. 1982). Fish contained from <0.1 to <0.5 ppm Cr. Higher levels were reported for bivalve molluscs (<0.1 to 1.5 ppm), gastropod molluscs (<0.1 to 1.4 ppm), and crabs (<0.1 to 0.9 ppm), depending upon location (Table 19). The Cr concentrations did not increase with presumed trophic level.

38. The concentrations of Cr in three saltwater food webs were studied using Cs:K ratios to assign the trophic levels (Young and Mearns 1979). Table 10 shows that Cr concentrations were similar in fishes from different trophic levels in the Salton Sea and Newport Bay food webs. In the Palos Verdes food web, the highest Cr levels were in molluscs and crabs, with lower levels in prawns and fishes. There was no tendency for Cr to biomagnify in

any of these food webs. A similar study was undertaken on a pelagic and an epibenthic food web (Schafer et al. 1982). In the coastal pelagic food web white sharks and zooplankton had 0.145 and 0.114 ppm Cr, respectively, with lower levels in all other organisms (Table 5). In the epibenthic Palos Verdes food web, sediments contained 802 ppm Cr, whereas the highest levels in the biota were 1.77 and 0.188 ppm, respectively, in mysids/decapods and prawns. There were no trends toward biomagnification of Cr in either food web.

#### Copper

39. Copper in freshwater organisms. Results from laboratory food chain work suggest that food may be a major source of copper (Cu) for aquatic organisms, but biomagnification of Cu probably does not occur. Cowgill (1976) studied the uptake of Cu by *Daphnia* spp. in an artificial food chain. Under these laboratory conditions, biomagnification did not occur. *Daphnia pulex* and *D. magna* periodically fed *Euglena gracilis* and mixed algal cultures contained similar or lower levels of Cu than did the algal cultures (Table 20). Both the algae and *Daphnia* were grown (separately) in spring water. No attempt was made to isolate uptake from food from bioconcentration from the water. In another food-chain study, Cu concentrations decreased with increasing trophic level (Tarifeño-Silva et al. 1982). Microcrustaceans (*Daphnia magna* and *D. pulex*) were fed exclusively on green algae (*Scenedesmus* spp.) grown in and harvested from Fe-enriched wastewater effluent. The *Daphnia* were then fed to the fish. The concentrations of Cu in algae, microcrustaceans, and fish were 200, 30 to 60, and 2.54 to 3.27 ppm, respectively (Table 1). A similar food-chain study was conducted using tubificid worms fed for 7 days exclusively on bacterial cultures previously exposed to Cu for 10 days (Patrick and Loutit 1976). The concentration of Cu in worms increased with Cu levels in the bacteria (Table 18). Worms fed on bacteria containing the lowest level (213 ppm) of Cu accumulated Cu to levels (236 ppm) only slightly exceeding those of the bacterial cultures. At higher bacterial-Cu concentrations, the Cu levels in the worms were about half those of the bacteria. The authors suggested that the worms treated at higher levels of bacterial Cu probably did not feed as much as those subjected to lower levels. This study indicates that bacteria are an important source of Cu for tubificid worms, but biomagnification of Cu did not occur during the 7-day feeding period.

40. Several field studies also suggest that Cu does not biomagnify in

freshwater food webs. Copper levels in the biota of the Illinois River (Table 14) decreased with increasing trophic level (Mathis and Cummings 1973). The average Cu level in detritivorous tubificid worms (23 ppm) was about the same as that of the sediments (19 ppm). The mean Cu levels ranged from 1.2 to 1.7 ppm in clams, 0.17 to 0.26 ppm in omnivorous fishes, and 0.07 to 0.19 ppm in predatory fishes. Cherry and Guthrie (1977) reported that the order of increasing Cu levels in a coal ash basin was water < plants < vertebrates < invertebrates < sediments (Table 13). Following dredging, Cu levels doubled in plants and invertebrates and tripled in vertebrates. The Cu levels in vertebrates were still about half those in invertebrates, however. No valid conclusion can be made regarding biomagnification, as the data were compromised by the inclusion of organisms that did not constitute a logical food chain. Anderson (1977) reported Cu levels in 35 genera of aquatic invertebrates from the Fox River in Illinois. The highest Cu concentrations reported were 95 and 99 ppm, respectively, for the crayfish, *Cambarus* sp., and the isopod, *Asellus* sp., both of which are largely detritivorous (Table 3). There was no obvious pattern of Cu concentrations, except that crustaceans, as a group, contained higher Cu levels than other groups, presumably because of a higher metabolic Cu requirement. Lewis (1980) also found no relationship between trophic level and Cu concentrations in the biota of four desert streams (Table 21). In streams with contaminated sediments, Cu levels were elevated at all trophic levels in comparison to levels in biota from less contaminated streams. Highest Cu levels were found in plants and insects, with lower levels in the vertebrates, except at Seven Springs Wash.

41. Copper in marine organisms. In the laboratory, Boothe and Knauer (1972) found that the concentration of Cu in feces of crabs fed on algae containing Cu was about four times that in the algae consumed. The apparent retention of Cu was about 40 percent of the total ingested, suggesting that diet is the primary source of Cu in marine crabs (Table 1). Another study (Rice, Tenore, and Windom 1981) demonstrated a decrease in Cu in the polychaete worm, *Capitella capitata*, as detrital ration was increased. Initial Cu concentrations in *Capitella*, *Ascophyllum nodosum* detritus, and *Gracilaria foliifera* detritus were 140, 9.6, and 9.7 ppm, respectively. Worms fed *Ascophyllum* detritus rations of 0.04 and 0.09 g N m<sup>-2</sup> day<sup>-1</sup> had average Cu concentrations of 2,100 and 1,400 ppm, respectively. Those fed *Gracilaria* detritus rations of 0.11 and 0.13 g N m<sup>-2</sup> day<sup>-1</sup> contained 2,200 and 1,500 ppm Cu, respectively.

There was wide variation in Cu content of the worms, however, and the apparent decreases in Cu with increased level of ration were not statistically significant. These data suggested that food would be a major source of Cu for marine organisms.

42. Field research largely has indicated that biomagnification of Cu does not occur in the marine environment. Surveys conducted by Preston et al. (1972) revealed no tendency toward Cu enrichment in herbivores. The range of Cu concentrations in the limpet, *Patella* sp., was similar to those in the brown alga, *Fucus* spp., and the red alga, *Porphyra* sp., collected at the same area (Table 4). As the authors indicated that the limpets were collected on rocks, it is more likely that the limpets were feeding on organisms on the surfaces of the rocks, rather than on the algae. Stickney et al. (1975) surveyed fishes and crustaceans in a Georgia estuary and found no relation between trophic levels and Cu concentrations in tissues. The highest Cu levels that were found in fish feeding on decapods averaged only 2 to 14 percent of those of the various decapods (Table 22). The highest average Cu concentrations in fish (2.3 ppm) were in those species feeding upon mysids and/or copepods. These data are the concentrations of Cu on a whole-organism basis in the crustaceans and in muscle tissue of the fish and, consequently, may not be truly representative of Cu levels in the fish. Phillips et al. (1982) reported that the ranges of Cu concentrations in fish, bivalve molluscs, cephalopod molluscs, and crustaceans from Victoria Harbor in Hong Kong were <0.1 to 1.1, 2.1 to 5.3, 4.4, and 1.1 to 35.2, respectively (Table 19). Copper levels in marine organisms from Card Sound, Florida, were reported by Gilio and Segar (1976). Table 6 shows that the highest mean Cu concentrations were 21, 12, 7.4, and 5.8 ppm, respectively, in epiphytes on *Thalassia*, *Laurencia* and phytoplankton, detritivores and carnivores combined, and dead mangrove (*Rhizophora mangle*) leaves. Sponges, which filter plankton and detritus particles from the water, contained 3.7 ppm Cu. These data indicate that food is an important source of Cu for marine animals, but biomagnification of Cu is unlikely.

43. Several studies of the movement of Cu within food webs of the Loire Estuary, France, have been reported. Amiard et al. (1980) found elevated levels of Cu in the gut contents of fishes in comparison with levels in the tissues (Table 8). These fishes had fed largely upon mysid crustaceans and polychaete worms. Prey species contained substantially greater Cu levels than did the predators (gut contents excluded). Crustaceans feeding on oligochaete

and polychaete worms had the highest Cu levels for whole organisms (including gut contents). A second study by the same group (Amiard-Triquet et al. 1980) indicated that, during the digestive process, the concentration of Cu decreased slightly in stomach contents and increased in the intestinal contents. The predators were planktivorous fish feeding largely upon copepods and mysids. The Cu levels in the prey were significantly greater than those in the predators, suggesting that Cu does not biomagnify. Metayer et al. (1980) also reported that Cu concentrations decreased with increasing trophic levels (Table 23). In this study, the highest average Cu levels were 4.1 and 4.8 ppm in fish feeding on zooplankton. All three of these studies suggest that food is an important source of Cu in estuarine fishes, but there was no evidence for biomagnification of Cu. Talbot and Chegwiddden (1982) examined Cu levels in molluscs, polychaetes, and crabs from Cockburn Sound in Australia. Table 7 (data from four sites) shows that the concentration of Cu in these animals varied rather widely with location. The variability of Cu levels within each species precluded any interpretation of possible trophic movement of Cu. Copper concentrations in the flesh of crabs fell within the same range as those in seagrasses (*Posidonia* spp.) and sea lettuce (*Ulva lactuca*) from other sites in Cockburn Sound.

44. Using Cs:K ratios to establish trophic levels, Young and Mearns (1979) and Schafer et al. (1982) studied marine webs off the coast of southern California. Without exception, the Cu concentrations decreased at the higher trophic levels (Tables 10 and 11). The highest Cu levels generally occurred in crustaceans, with the exception of the 15.8 ppm in squid in a coastal pelagic food web (Table 10). Polprasert (1982) also reported Cu levels in marine fishes ranked by trophic level, but did not indicate how trophic level was determined and only reported data for levels III and IV. The maximum value reported for trophic level III (21.59 ppm) exceeded that for level IV (3.54), but the ranges overlapped greatly. Generally, Cu concentrations reported by Polprasert for level III equalled or exceeded those for level IV, which suggests that Cu does not biomagnify.

45. Although most of the field evidence indicates that Cu does not biomagnify in marine food webs, a few studies suggest that biomagnification of Cu may occur over short links of the food chain, particularly in molluscs and in seabirds. Kay and Rojanivipart (1976) reported Cu concentrations in periwinkles (*Littorina littorea*) which greatly exceeded those in macrophytic

vegetation and sediments taken from the same locations in a saltmarsh of the Burry Inlet. The levels of Cu reported in vascular macrophytes, detritus, macroalgae, sediments, and periwinkles were 5 to 13, 5 to 25, 8, 13 to 52, and 130 to 147 ppm, respectively, in areas from which periwinkles were collected. Although these data suggest that biomagnification of Cu may occur in the herbivorous periwinkle, they are inconclusive, since no data were provided for Cu levels in epiphytes, the primary food of periwinkles, and the data do not account for possible direct absorption through the foot. A similar pattern for Cu was reported by Jenkins (1976). Enrichment factors (compared with seawater) for molluscs, other invertebrates, algae, fish, and zooplankton were 350, 20, 10, 7.5, and 4.9, respectively. These data suggest that biomagnification of Cu may occur at the lower levels of the food web, particularly in molluscs and invertebrates, but not at the uppermost levels. Data reported by Greig et al. (1977) also suggest that Cu is not biomagnified through the upper trophic levels (Table 16). The Cu concentrations in flesh of whelks, crabs, and flounders at Long Island Sound (Area 2) averaged 21, 13, and 1.4 ppm (wet weight), respectively. These levels may possibly reflect the respective metabolic requirements for Cu in those species. High concentrations of Cu were found in deposit-feeding clams, periwinkles, and dogwhelks (Bryan and Hummerstone 1977). Table 9 shows mean Cu concentrations >100 ppm in all of these organisms. Limpets (*Patella vulgata*) and dogwhelks (*Nucella lapillus*), which feed on limpets, averaged 19 and 110 ppm Cu, respectively. The deposit-feeding clam, *Macoma balthica*, averaged 300 ppm Cu (range 96 to 615 ppm). The data do not generally support biomagnification of Cu, except possibly from limpets to dogwhelks.

#### Lead

46. Lead in freshwater organisms. Several laboratory studies have investigated the trophic movement of lead (Pb) in freshwater organisms. Patrick and Loutit (1976) examined the passage of Pb from wastewater effluents through bacteria to tubificid worms and concluded that bacteria may be an important intermediary between the physical environment (effluent) and the aquatic food webs. Tubificid worms (*Tubifex* sp. and *Limnodrilus* sp.) fed exclusively on bacteria (*Sphaerotilus* sp.) containing 119, 410, and 721 ppm Pb, respectively, accumulated 179, 559, and 568 ppm Pb (Table 18). The differences between the initial Pb concentration (151 ppm) in the worms and the concentrations after 7 days of feeding, followed by 1 day without food (to allow for evacuation of



gut) suggested that increased body burdens of Pb came from food. Accumulation of Pb from water or via Pb leached from the bacteria were not considered. In a similar study, Cowgill (1976) fed microcrustacea on *Euglena gracilis* and mixed algal cultures, harvesting weekly for 3 months. The concentrations of Pb in spring water, algae, and microcrustaceans were 0.85 ppb, 10.0 to 12.6 ppm, and 7.2 to 9.8 ppm, respectively (Table 20). The authors concluded that the composition of the microcrustacean was "governed by the mixed algal culture and the spring water." Biomagnification of Pb obviously did not occur in this system, and the relative importance of bioconcentration and trophic transfer of Pb could not be determined. Tarifeño-Silva et al. (1982) added fish to a similar algae-*Daphnia* system to provide a three-level trophic chain. The order of decreasing Pb concentration was algae > *Daphnia* sp. > fish muscle >> water (Table 1). *Daphnia magna* accumulated significantly higher levels of Pb than *D. pulex*. The magnitude of direct uptake of Pb from the water by either *Daphnia* or fish was not determined. Biomagnification of Pb apparently did not occur in any of these laboratory studies, at least at high levels of exposure.

47. Analyses of field-collected organisms also indicate that biomagnification of Pb is unlikely in freshwater food webs. The values of mean Pb concentrations in the Illinois River (Mathis and Cummings 1973) were 28, 17, 2.2 to 3.7, 0.56 to 0.84, and 0.34 to 0.98 ppm, respectively, for sediments, tubificid worms, clams, omnivorous fishes, and predatory fishes, and only 2 ppb in the water (Table 14). Anderson (1977) showed no trophic-level relationship for Pb in 35 genera of invertebrates from the Fox River. The Pb concentrations were higher in benthic invertebrates, and lower in the free-swimming aquatic insects (Table 3). Predators (such as the leech, *Placobdella*, and many aquatic hemiptera) often had substantially lower Pb levels than the detritivores and filter-feeders. Enk and Mathis (1977) also found no significant differences between Pb concentrations in predatory and non-predatory species (Table 2). The general order of increasing Pb concentrations was water < fishes < aquatic insects < sediments < snails. The highest levels observed were 13.6 and 12.6 ppm (wet weight), respectively, in a snail (detritivore) and a damselfly (predator). Data collected as part of the National Pesticide Monitoring Program showed no trend toward biomagnification of Pb in freshwater fishes (May and McKinney 1981).

48. Lead in marine organisms. Most of the data on Pb in marine food

webs come from analyses of field-collected materials and suggest that Pb is not biomagnified. A laboratory study by Boothe and Knauer (1972) implied that food may constitute an important source of Pb accumulation, however. Crabs fed algae containing 4.3 ppm Pb ingested a total of 25 µg Pb/g fecal material produced (Table 15). The Pb in feces accounted for about 75 percent of the total ingested Pb. This suggests that part or all of the remaining 25 percent of ingested Pb may have been retained in the crabs. The authors did not give data on Pb levels within the crabs' tissues.

49. Jenkins (1976) made an extensive survey of the literature and calculated "maximum enrichment factors" for Pb in marine organisms, based upon the levels reported in the literature as compared with a mean Pb level of 0.04 ppb in seawater. The enrichment factors ( $\times 10^3$ ) for Pb were 500, 147, 375, 1,000, and 75, respectively, for algae, zooplankton, invertebrates except molluscs, molluscs, and fish.

50. Several field studies reported Pb in various marine biota. Analyses of algae and limpets collected in coastal waters of the British Isles indicated elevated Pb levels in a first-level consumer (Preston et al. 1972). In two areas, Pb concentrations in the limpet, *Patella* sp., were about double those in marine macroalgae, *Fucus* sp. and *Porphyra* sp. (Table 4). Whether the limpets actually fed on the algae or, more likely, grazed upon epilithic or epiphytic organisms was not indicated. Consequently, biomagnification cannot be positively demonstrated. A study of trace elements in Card Sound, Florida, showed no trend toward biomagnification of Pb (Gilio and Segar 1976). The mean Pb concentrations in the macrofauna were less than or equal to those in phytoplankton, epiphytes on seagrasses, and macrophytic vegetation (Table 6). Greig et al. (1977) reported substantially higher Pb levels in crabs than in whelks at a single location (Long Island Sound--Area 2), but Pb decreased by an order of magnitude from crab to flounder (Table 16). At the Chincoteague Inlet site, Pb concentrations were lowest in clams and crabs (<0.5 ppm) and approximately the same in whelks and flounder (0.9 and 0.8 ppm, respectively). The data suggested that biomagnification of Pb probably did not occur. Talbot and Chegwiddden (1982) found that Pb in flesh of the crab, *Portunus pelagicus*, was less than or equal to those of molluscs collected within the same areas (Table 7). Lead in the hepatopancreas of the crabs was two to four times that in the flesh, however. Analyses of finfish and shellfish from Victoria Harbor, Hong Kong, showed no trend toward biomagnification of Pb (Phillips et al.

1982). Table 19 shows that the highest levels observed were in bivalve molluscs (0.1-3.0 ppm), whereas the other invertebrates and finfish had lower Pb levels (<0.1-0.3 ppm).

51. Several studies have included feeding habits along with tissue analyses. Stickney et al. (1975) found no substantial differences in the Pb content of either fish or invertebrates and no correlation between food habits and Pb content of fishes in a Georgia estuary (Table 22). Lead levels in the Looe Estuary (England) showed no tendency to biomagnify within the invertebrate community (Bryan and Hummerstone 1977). The highest levels were observed in a deposit-feeding clam, *Scrobicularia plana*; lowest levels were in the predatory dogwhelk, *Nucella lapillus* (Table 9). The Pb concentration in dogwhelks was about 20 percent of that in limpets, a common food of the dogwhelk.

52. Analyses of the fauna of the Loire Estuary (France) provided no evidence for the biomagnification of Pb in estuarine food webs. Amiard et al. (1980) showed that Pb concentrations in predators were usually much lower than those either in live prey or in the gut contents (Table 8). Trophic transfer factors  $\frac{\text{concentration in predator}}{\text{concentration in prey}}$  were <1 for Pb in plantivorous fishes

(Amiard-Triquet et al. 1980). Metayer et al. (1980) observed that the range of Pb was similar in both omnivorous and carnivorous fishes (Table 23). Analyses of fish stomach contents showed significantly higher levels of Pb than observed in the fishes.

53. The use of Cs:K ratios to assign trophic level also has shown that Pb does not biomagnify within marine food webs. Young and Mearns (1979) reported similar levels of Pb in fish representing different trophic levels of the Salton Sea and Newport Bay food webs (Table 10). In the Palos Verdes food web, these authors also reported similar Pb levels at all trophic levels, with the exception of scorpionfish. The scorpionfish represented the top trophic level in the food web and contained Pb concentrations an order of magnitude greater than those in most of the other species, including bocaccio, which also was assigned to the same trophic level as the scorpionfish. Schafer et al. (1982) also found no evidence of Pb biomagnification (Table 11). The highest levels of Pb were in the zooplankton, and the lowest were in the white shark (top predator) and squid (intermediate trophic level) in a coastal pelagic food web. Mysids and decapods had the highest Pb contents in an

epibenthic food web. Polprasert (1982) also ranked fishes in the Chao Phraya River Estuary according to trophic level. Based upon the average Pb concentrations in many samples, he concluded that biomagnification of Pb might possibly occur. The average Pb concentration in 69 samples from level III and 71 samples from level IV were 11.56 and 13.19 ppm, respectively. The ranges of the mean Pb concentrations for different species at levels III and IV were 0.49 to 283.91 and 0.08 to 35.5 ppm, respectively. The author did not include representatives of other trophic levels or specify how trophic levels were determined. Consequently, it was difficult to conclude biomagnification of Pb.

Mercury

54. Mercury in freshwater organisms. The uptake and movement of mercury (Hg) within freshwater food chains have been studied intensively in the laboratory. Lock (1975) compared the uptake of methylmercury from food and water by *Daphnia pulex* and rainbow trout (*Salmo gairdneri*). For the food-chain studies, the *Daphnia* were fed algal cells (*Chlamydomonas reinhardtii*) that had been previously exposed to various levels of methylmercury; trout were fed pelleted food containing methylmercury. Table 24 shows the uptake of methylmercury from food and water. Methylmercury apparently did not biomagnify in this laboratory study. The data were difficult to compare, however, as Hg in fish was reported on a fresh-weight basis, whereas that in the diet was reported as parts per million dry weight. The author concluded that the uptake of methylmercury was more rapid from water than from food, but the "percentage uptake of mercury was 5-10 times higher from the latter source." The author stated further that most of the methylmercury burden of aquatic organisms would come from food, since most methylmercury is complexed with organic matter and dissolved methylmercury is present in water at concentrations lower than 1 pptr (parts per trillion).

55. A series of studies using three- and four-step trophic chains (algae-*Daphnia*-mosquitofish-trout) indicated the importance of food as a source of Hg. Using "global contamination" (i.e., simultaneous contamination via both water and food) with methylmercury, Boudou et al. (1979) found that direct methylmercury uptake at 18° C was about half that of global uptake. Global accumulation and direct accumulation were similar at 10° C, however. Temperature had a pronounced effect only on global uptake, presumably as the result of increased feeding at the higher temperatures. The concentration of Hg in the water was 1 ppb, but no information was given about the organismal

Hg concentrations in the algae or *Daphnia*. Ribeyre, Boudou, and Delarche (1979) added trout, *Salmo gairdneri*, to the trophic chain and similarly found substantially greater methylmercury uptake via global contamination than from water alone. Global Hg uptake was less at 26° C than at 18° C, probably the result of greater feeding by trout at the cooler temperatures. This temperature effect was again demonstrated by Ribeyre, Delarche, and Boudou (1980). The major difficulty in interpreting the results of these three studies is that, using global contamination, all components of the system were present simultaneously. No studies were done to demonstrate trophic contamination and trophic movement of methylmercury in the flow-through systems using uncontaminated water.

56. A similar problem arises in the study by Cowgill (1976), in which algae were cultured in spring water and then fed separately to *Daphnia* sp., which were also being raised in spring water. The concentration of total Hg in the *Daphnia* was 30 to 40 percent less than that in the algae (Table 20). The water contained 0.13 ppb Hg, in contrast to 4.2 to 4.25 ppm in the algae and 2.68 to 3.30 ppm in *Daphnia*. The source of Hg in *Daphnia* remains uncertain.

57. Phillips and Buhler (1978) addressed the problem of methylmercury accumulation from food and water. Uptake of methylmercury by trout over a 24-day period was essentially linear, regardless of whether the route of exposure was water, food, or both. Uptake from food and water was additive, and accumulation from one source had no influence on that from the other (Table 25). The authors indicated that about 70 percent of the methylmercury ingested and 10 percent of that passed over the gills was assimilated by the trout. This study suggests that methylmercury uptake from food might result in biomagnification. The evidence was not conclusive, however.

58. Bacteria are important in the movement of Hg into aquatic food webs because of their involvement in the conversion of inorganic Hg into organic methylmercury. Hamdy and Prabhu (1979) used radioactive  $^{203}\text{Hg}$  to follow the movement of Hg into bacteria. Bacteria (*Bacillus licheniformis*) were exposed to 110 ppb  $^{203}\text{Hg}$  as  $^{203}\text{Hg}(\text{NO}_3)_2$  or  $\text{C}_6\text{H}_5^{203}\text{HgOOCCH}_3$ . The ratios of organic to inorganic  $^{203}\text{Hg}$  absorbed by bacteria were 4.67, 4.63, and 7.92, respectively, when incubated 96 hr at 23°, 37°, and 45° C. When the bacteria were fed for 7 days to mosquito larvae (*Aedes aegypti*), the larvae absorbed about 25 percent as much  $^{203}\text{Hg}$  from bacteria labelled with organic Hg as from those

labelled with inorganic Hg. When larvae labelled with organic or inorganic  $^{203}\text{Hg}$  were fed to guppies (*Lebistes reticulatus*), however, guppies accumulated similar levels of  $^{203}\text{Hg}$  from both sources. The same was observed for cichlid fish (*Cichlasoma facetum*) feeding on labelled guppies. The concentration factors shown for both organic and inorganic  $^{203}\text{Hg}$  in the cichlids was substantially  $>1$  only in the gut and in feces, and  $<1$  in guppies. These data showed that food was an important source of Hg, particularly in the primary consumer. Analyses for  $^{203}\text{Hg}$  were done by scintillation counting, with no mention of the forms of Hg which the consumers actually ate. The trophic movement of  $^{203}\text{Hg}$  was compromised by labelling the food organisms by direct contamination, rather than by passage of  $^{203}\text{Hg}$  through a continuous trophic chain, however, and no definitive conclusions could be made from this study regarding biomagnification. Longer term feeding studies may be necessary to clarify the role of food in Hg movement to the secondary and tertiary consumers in such laboratory microcosms.

59. The movement of Hg through aquatic food webs has been studied in more detail under field conditions. Cherry and Guthrie (1977) measured Hg levels in the components of a coal ash basin, both before and after dredging. Table 13 shows similar Hg levels in all the biotic components of this system. The averaging of Hg levels in dissimilar types of organisms and the inclusion of organisms that do not constitute a logical food chain compromise the value of the study. The data do suggest a lack of biomagnification of Hg in this system, however.

60. Potter, Kidd, and Standiford (1975) demonstrated a relationship between trophic level and Hg content of organisms from Lake Powell, Arizona. Figure 1 shows clearly that Hg was biomagnified by fish in this study. The highest Hg levels were in the top predators (walleye and bass). The relatively high levels in carp and catfish are likely the result of an omnivorous feeding habit. Paasivirta et al. (1983) also examined Hg in fish. At three lakes in Finland, the mean Hg levels were higher in pike than in roach. The data were quite variable, however. Biomagnification of Hg may have occurred at Paijanne, where Hg levels in roach and pike were  $0.238 \pm 0.107$  and  $0.660 \pm 0.095$  ppm, respectively. Most (85-95 percent) of the Hg in both species was methylmercury. Possible evidence of Hg biomagnification also was shown by Hildebrand, Strand, and Huckabee (1980). Fishes (rock bass and hogsucker) in the North Fork Holston River contained higher mean levels of Hg than did

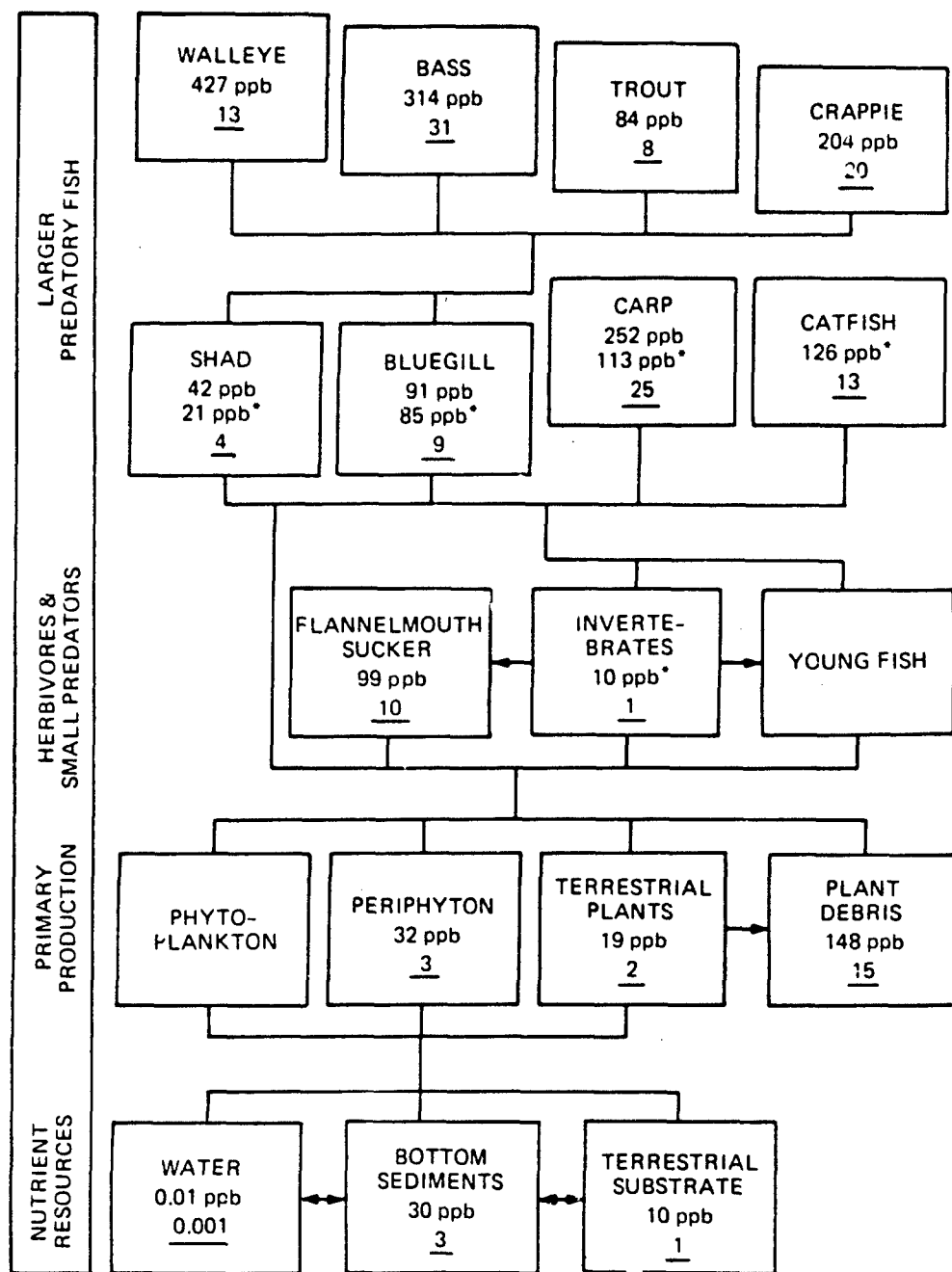


Figure 1. Relationship of Lake Powell trophic levels to mean parts per billion mercury and magnification factor (underlined) relative to the average concentration of the terrestrial sandstone substrate. Magnification factor based on water would be 1000 times greater. Values with an asterisk are whole body analyses; other fish values are of axial muscle (from Potter, Kidd, and Standiford 1975). Reprinted with permission from *Environmental Science and Technology*, Vol 9, p 44, Copyright 1975--American Chemical Society

invertebrates (Table 26). The authors reported that the majority of the Hg in fishes at all sampling stations was methylmercury.

61. Several other studies have examined Hg levels in freshwater fishes from different locations. Fimreite and Reynolds (1973) reported higher levels of Hg in the top predators than in fish at lower trophic levels in mercury-contaminated areas. Suckers (omnivore/detrivore) had lower Hg than pike, walleye, and burbot (Table 27). The Hg level in muscle had a high positive correlation with body weight. An earlier paper (Fimreite et al. 1971), however, showed no clear trend toward increasing Hg concentrations at higher trophic levels and no apparent relationship between body weight and Hg levels in fish (Table 28). Henderson and Shanks (1973) compared Hg concentrations in fishes from the Columbia River system and from streams in other areas of the United States, including Alaska and Hawaii. In this study, there was no apparent correlation between Hg level and body weight. The highest Hg levels were found in the northern squawfish, a predatory species collected in the Columbia River system, but not elsewhere. Akielaszek and Haines (1981) attributed the levels of Hg in the top predators in Eagle and St. Froids Lakes to the presence of rainbow smelt, an abundant forage species in these lakes. In Cliff Lake, which is similar to the other two lakes both physicochemically and biologically but lacks rainbow smelt, Hg levels in the predators were lower than in Eagle and St. Froid Lakes (Table 29). There was no correlation between fish size and Hg content. May and McKinney (1981) also reported substantially higher Hg levels in predators than in other fishes from locations throughout the United States and Canada. Their data also showed no relationship between weight and Hg accumulation. From the data presented in these papers, it appears that Hg may be biomagnified in the top predatory fish species, especially in regions of known Hg contamination.

62. Mercury in marine organisms. Bacteria have an important role in the mobilization and entry of Hg into the marine food web. The presence of Hg-reducing or Hg-accumulating bacterial strains of *Pseudomonas* sp. greatly enhanced the absorption of  $^{203}\text{Hg}$  from seawater by the oyster, *Crassostrea virginica* (Sayler, Nelson, and Colwell 1975). In the presence of these bacteria, the majority of  $^{203}\text{Hg}$  in the water was associated with the particulate (bacterial) fraction. After 4 days incubation, the concentrations of Hg were significantly greater than the controls in whole oyster, gills, and adductor muscle when Hg-accumulating bacteria were present and in the gills when



Hg-reducing bacteria were present (Table 30). Similar studies by Colwell et al. (1975) indicated  $^{203}\text{Hg}$  accumulation by bacterivorous ciliates. After 4 hr of incubation, about 20 percent of the  $^{203}\text{Hg}$  activity localized in bacteria had transferred to the ciliates. Further work by Berk and Colwell (1981) demonstrated the transfer of  $^{203}\text{Hg}$  from labelled bacteria (*Vibrio* sp. and *Pseudomonas* sp.) through ciliates (*Uronema nigricans*) to a copepod (*Eurytemora affinis*). The concentrations of Hg in ciliates fed on bacteria containing 6.6, 14, and 50-60 ppm Hg were 86, 200, and 160-340 ppm, respectively. The authors indicated that 44-53 percent of  $^{203}\text{Hg}$  present in ciliates was retained by copepods fed on the ciliates. The Hg concentration in ciliates fed to the copepods was not shown, but the authors stated that Hg was not biomagnified from ciliate to copepods in 72-hr feeding experiments.

63. Trophic transfer of Hg from jack mackerel to yellowtail was reported in a laboratory study (Suzuki and Hatanaka 1974). Yellowtail were fed on jack mackerel previously raised in methylmercury-dosed seawater. The mean Hg level in yellowtail increased from 0.05 to 0.43 ppm during an 18-day feeding period. The level of Hg in jack mackerel used as the diet for yellowtail was not specified, but the authors indicated that the transfer of Hg from diet to yellowtail averaged 88 percent. This suggests that food is an important source of Hg in the marine food chain.

64. Several studies with field-collected marine organisms have suggested that Hg is not always biomagnified. Polprasert (1982) ranked fish in Chao Phraya River Estuary into either trophic level III or IV and found no difference in the range of Hg concentrations. The ranges of Hg concentrations at levels III and IV were 0.01 to 0.70 and 0.01 to 0.57, respectively. Average Hg concentrations for these two levels were 0.10 (n=69) and 0.16 (n=71) ppm, respectively. Polprasert's contention that this represented "possible bio-magnification" is difficult to reconcile from his data without further information on feeding habits and without data on Hg at lower trophic levels. How trophic levels were determined was not indicated. Leatherland et al. (1973) also have shown poor correlation between Hg concentration and trophic level. The shark (*Etmopterus spinax*), which was designated the top predator in the system, had lower Hg levels than other species except the jellyfish, *Pelagia* sp., and a tunicate, *Pyrosoma* sp. (Table 5). Greig, Wenzloff, and Shelpuk (1975) also found no obvious correlation between feeding habit and Hg content of fish in the North Atlantic (Table 31). The mean Hg concentrations

in fish muscle and fish livers of 41 species of fish were 0.154 ( $\pm$  0.124) and 0.164 ppm, respectively. Invertebrates usually had <0.1 ppm Hg. Levels in plankton, pandallis shrimps, scallops, squid, and sediments usually were <0.05 ppm. One lobster sample had 0.31 and 0.60 ppm Hg in tail muscle and liver, respectively. Another study by Greig et al. (1977) showed that Hg did not biomagnify in benthic biota (Table 16). The Hg concentrations in flounder were approximately the same as those in crabs. Stoeppler et al. (1979) also reported similar results for benthic organisms. Phillips et al. (1982) reported levels of Hg in finfish, bivalve molluscs, cephalopods, crabs, and shrimps taken from Victoria Harbor, Hong Kong, to be <0.1 to 0.3, 0.1 to 0.5, <0.1, <0.1 to 0.3, and <0.1, respectively (Table 19).

65. Most of the field-collected data suggest that Hg is biomagnified in marine food webs, however. Field data relating feeding habits or trophic level to Hg content seem to confirm that biomagnification of Hg often occurs in marine organisms. Jenkins (1976) reported maximum relative Hg concentration factors of 0.8, 8.4, 10.0, and 31.0, respectively, for marine algae, invertebrates except molluscs, molluscs, and fish. Ratowsky, Dix, and Wilson (1975) found Hg levels >0.5 ppm in 51 percent of the fish which feed predominantly on other fish, in contrast to only 24 and 7 percent, respectively, in invertebrate-feeding and herbivorous fish (Table 32). Stickney et al. (1975) showed higher Hg levels in fish feeding upon mysids and decapods than in the prey organisms (Table 22). Marine organisms ranked into trophic levels on the basis of Cs:K ratios also showed increased Hg concentrations at the upper trophic levels (Young and Mearns 1979). Both organic and total Hg were magnified in the top predators in the Salton Sea and at Newport Bay (Table 10). Similar results were reported by Schafer et al. (1982) in open ocean, coastal pelagic, and epibenthic food webs off California (Table 11).

66. Along the coast of the southeastern United States large predators, especially sharks, contained significantly higher levels of methylmercury and total Hg (Table 33) than the smaller forage species, such as the menhaden and killifishes and the crustaceans (Gardner et al. 1975). Klemmer, Unninayer, and Okubo (1976) reported average Hg concentrations of 0.26, 0.10, 0.08, and 0.03 ppm, respectively, in secondary benthic carnivores, primary benthic carnivores, omnivores, and herbivores from the coastal waters of Hawaii. When separated by benthic feeding habit, the average Hg concentrations in herbivores, omnivores, and carnivores, respectively, were 0.022, 0.058, and

0.075 ppm in species feeding in direct contact with the sediment, and 0.036, 0.070, and 0.080 ppm in species feeding above the sediment-water interface. Matsunaga (1978) reported that Hg accumulation probably "depends on food chain amplification." Table 34 shows high Hg levels in many of the large predatory species, particularly tuna, skipjack, and seabass, and in the rockfishes (*Sebastes* spp.), and low Hg levels in small forage species and invertebrates. Similar results were reported for fish and shellfish caught off the Indian coast (Ramamurthy 1979). Figure 2 shows that the top predators (tuna and shark) had substantially higher mean Hg levels than did detritivores (shrimps), filter-feeders (oysters and clams), plankton-feeders (mackerel), and small forage species (sardines and ribbonfish). Similar results were reported by Stoeppler et al. (1979) for marine pelagic species from the Atlantic Ocean and the Mediterranean Sea. The top predators, swordfish (*Xiphias gladius*) and horse mackerel (*Trachurus trachurus*), contained 1,260 and 1,225 ppb Hg, respectively (Table 35). Significantly lower levels were found in small forage species, such as anchovy (*Engraulis encrasicolus*). Hilmy, Shabana, and Saied (1981) reported that the mean Hg concentrations in fish (*Aphanius dispar*), shrimp (*Sergestes lucens*), and mussels (*Modiolus modiolus*) taken from the Red Sea were 0.50, 0.25, and 0.33 ppb, respectively.

#### Zinc

67. Zinc in freshwater organisms. Several laboratory studies have indicated that zinc (Zn) does not biomagnify in aquatic systems. Cowgill (1976) fed microcrustaceans (*Daphnia* spp.) on algae cultured in spring water. The Zn levels in the algae and in the crustaceans following the feeding period were 304 to 339 ppm and 102 to 135 ppm, respectively (Table 20). In another investigation, the Zn concentrations in tubificid worms fed upon bacteria grown in wastewater effluent increased in direct proportion to Zn levels in the bacteria (Patrick and Loutit 1976). The Zn concentrations in the worms were substantially lower than those in the bacteria, except at the lowest level of bacterial Zn (Table 18). This study showed that bacteria were an important source of Zn for tubificids, but biomagnification did not occur. A similar study (Tarifeño-Silva et al. 1982) used a three-step trophic chain including algae, microcrustaceans, and fish. Table 1 shows that the Zn concentrations decreased from algae to *Daphnia* spp. to fish.

68. Another study suggested both that food was an important source of Zn and that Zn might be biomagnified, at least within the lower levels of

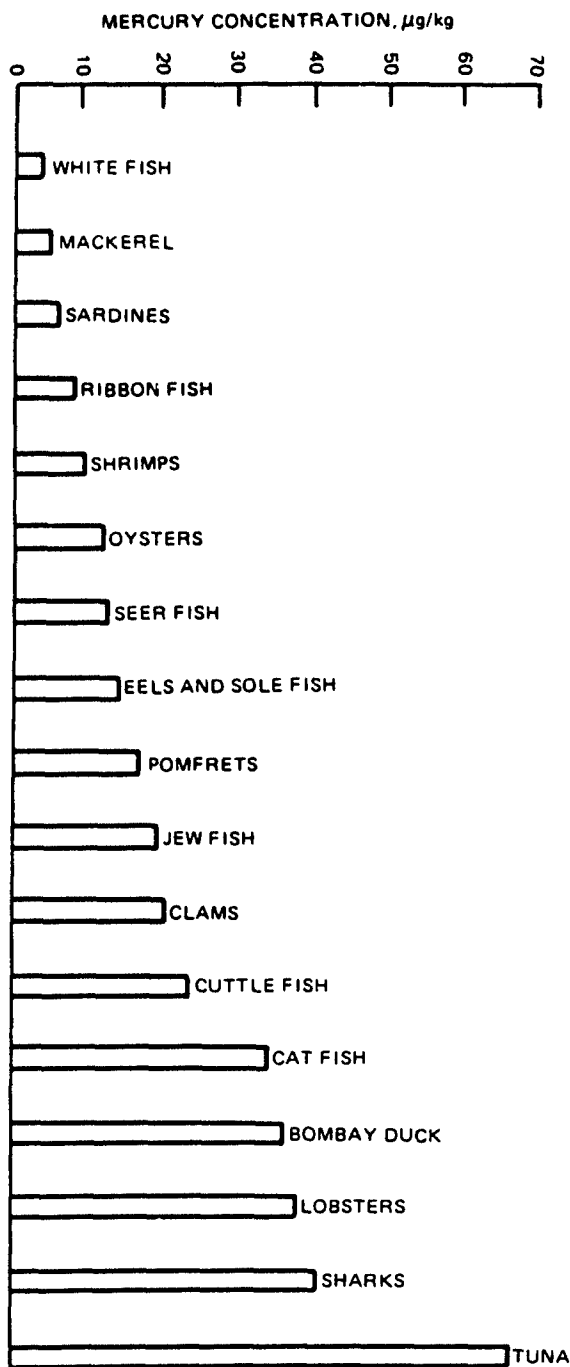


Figure 2. Mean mercury concentrations in marine food fishes and shellfish caught off the Indian coast (from Ramamurthy 1979). Reprinted from the Bulletin of the Japanese Society of Scientific Fisheries, Vol 45, p 1406, Copyright 1979--Japanese Society of Scientific Fisheries

aquatic food chains (Elwood, Hildebrand, and Beauchamp 1976). Aquatic fly larvae (*Tipula* sp.) fed on leaf detritus contained Zn concentrations about threefold those of the leaf detritus (Table 12), even after gut evacuation.

69. Evidence from field studies suggests that Zn concentrations in representatives of the upper trophic levels rarely exceed and frequently are much lower than those of their food organisms. Mathis and Cummings (1973) found the highest Zn levels in clams and detritivorous annelids (tubificid worms) and substantially lower levels in fishes (Table 14). Similar Zn concentrations were present in both predatory and omnivorous fishes. Anderson (1977) reported a wide range of Zn concentrations for 35 genera of invertebrates from the Fox River, Illinois. The highest levels were generally found in detritus- and sediment-dwelling organisms (e.g., clams, decapods, caddisflies, and mayflies), which are mostly detritivorous or filter-feeding species (Table 3). Lower Zn levels were found in the pelagic predators (bugs and beetles) and in the benthic predators (dragonflies and leeches). A notable exception was the predatory giant waterbug, *Belostoma* sp., which contained 228 ppm Zn. Lewis (1980) showed generally higher Zn levels in aquatic insects than in aquatic vertebrates, but the relative positions of the groups of organisms with respect to Zn concentrations varied somewhat with location (Table 21). Cherry and Guthrie (1977) indicated somewhat higher Zn concentrations in animals than in plants within a coal ash basin, both before and following dredging operations (Table 13). Before dredging, average Zn concentrations in invertebrates were slightly greater than those in vertebrates. Following dredging, this trend reversed. It is difficult to interpret the data as biomagnification, however, as the organisms analyzed did not constitute a logical food chain.

70. Zinc in marine organisms. Radioactive  $^{65}\text{Zn}$  has been used to follow the uptake of Zn from food and water in laboratory studies. Renfro et al. (1975) used a global contamination design to examine the relative importance of food and water as pathways for entrance of Zn into marine food chains. The specific activity of  $^{65}\text{Zn}$  increased in shrimp (*Lysmata seticaudata*) immediately following feeding on brine shrimp (*Artemia salina*) and then decreased within 24 hr to levels similar to those of shrimp fed upon uncontaminated brineshrimp and which received  $^{65}\text{Zn}$  solely from the water. The source (food or water) of the  $^{65}\text{Zn}$  for *Artemia* had no influence upon  $^{65}\text{Zn}$  uptake by the shrimps. Crabs (*Carcinus maenas*) absorbed more  $^{65}\text{Zn}$  globally (simultaneous exposure via both food and water) than from water alone, but lost the

additional  $^{65}\text{Zn}$  during molting. Fish (*Gobius* sp.) accumulated significantly more  $^{65}\text{Zn}$  from food (*Artemia* and mussels, *Nytilus galloprovincialis*) and water than from water alone. The data did not indicate biomagnification of  $^{65}\text{Zn}$  over a 90-day feeding period, but food did appear to be an important source of  $^{65}\text{Zn}$  activity. Similarly, Young (1977) reported that food was the major source of  $^{65}\text{Zn}$  for dogwhelks exposed to labelled seawater or labelled food (barnacles, *Balanus balanoides*). The author stated that  $^{65}\text{Zn}$  was not "accumulated up the food chain," however. Booth and Knauer (1972) also showed evidence that food is a major source of Zn. Table 15 shows that crabs fed on algae retained about 67 percent of the Zn ingested. The Zn concentration in crabs following feeding on the algae was not specified.

71. The majority of the field studies available reflected the results of laboratory investigations showing food as the major source of Zn for marine organisms, but did not show biomagnification of Zn within the food web. Jenkins (1976) surveyed the literature for heavy metals in marine organisms and calculated relative maximum enrichment factors of 13.8, 6, 57, 500, and 160, respectively, for algae, zooplankton, invertebrates except molluscs, molluscs, and fish. Preston et al. (1972) reported Zn levels in the limpet, *Patella*, that were within the ranges of Zn in macroalgae from the same area (Table 4). Leatherland et al. (1973) reported that the Zn concentration in a predatory fish (*Diaphus dumerili*) was substantially lower than those observed in pelagic crustaceans (Table 5). Gilio and Segar (1976) showed substantially lower Zn concentrations in the macrofauna of Card Sound, Florida, than in the phytoplankton, epiphytic algae, and macrophytes (Table 6). At various sites along the northeastern coast of the United States, Zn levels (Table 16) were elevated in crabs, in comparison with molluscs (Greig et al. 1977). The Zn levels in flounder were significantly less than those in the crabs, however. Talbot and Chegidden (1982) indicated that the Zn concentration in the flesh of crabs was about the same as that in polychaetes and mussels (Table 7). Substantially higher Zn levels were present in the hepatopancreas than in the flesh of the crabs. The range of Zn concentrations reported by Phillips et al. (1982) for finfish was considerably less than those for molluscs and crustaceans (Table 19).

72. Several reports that correlated trophic level or food habits with Zn levels also indicated that biomagnification of Zn does not occur in the marine environment. Stickney et al. (1975) found that Georgia estuarine

fishes contained Zn concentrations less than or equal to those in their primary foods (Table 22). Bryan and Hummerstone (1977) found no substantial differences in the concentrations of Zn in a variety of benthic invertebrates from the Looe Estuary. The highest average Zn levels were found in deposit-feeding clams, and lowest were in the cockle. The range of Zn concentration in the predatory dogwhelk widely overlapped that of one of its common food organisms, the limpet. Similar results have been obtained by using Cs:K ratios to rank organisms according to trophic level. Young and Mearns (1979) found no evidence of Zn biomagnification in any of three California saltwater food webs (Table 10). Shafer et al. (1982) also reported that Zn did not biomagnify either in a pelagic or an epibenthic marine food web off the coast of California (Table 11).

73. There was a suggestion of possible Zn biomagnification, however. Amiard-Triquet et al. (1980) reported that Zn was accumulated more from food than from water by planktivorous fish and that concentration factors were often  $>1$  for Zn. Their data, however, also showed that Zn levels in herring, sprat, and copepods field collected from the Loire Estuary were about 97, 67 to 100, and 225 to 228 ppm, respectively, suggesting no biomagnification. Consequently, the interpretation of their data as in any way supporting the biomagnification of Zn must be done with reservation. Another study by the same authors (Amiard et al. 1980) showed higher Zn levels in several species of fish than in their major food organisms (Table 8). The Zn levels in a fish designated by the authors as the "supercarnivore," as well as in several other fishes, were approximately equal to those in their food. Metayer et al. (1980) found that Zn in predatory and omnivorous fishes of the Loire Estuary often exceeded that of their prey by as much as a factor of 2. Table 23 shows that fish feeding on zooplankton and an annelid (*Boccardia ligERICA*) had substantially lower Zn levels than their food.

#### Summary

74. The information reviewed indicates that heavy metals do not biomagnify greatly either in freshwater or marine food webs, with the possible exception of methylmercury. Field and laboratory studies show that food may be an important source for the bioaccumulation of toxic heavy metals, particularly those which are essential trace elements (Cu, Zn, and Se), but also some which have no known metabolic functions (Cr, As, Cd, Hg, and Pb). These elements did not biomagnify to any extent within the food web, however.

Concentrations of these elements were generally higher in the tissues of benthic species (particularly herbivores and detritivores) and, occasionally, planktivores than in the top-level carnivores. In the case of Hg, laboratory evidence suggested that biomagnification would not occur, but was contradicted by the majority of the field studies, which indicated biomagnification. Methylmercury has an affinity for muscle and tissues and apparently is biomagnified through the trophic web to the top predators. Consequently, high concentrations of methylmercury are frequently found in the large, commercially valuable fishes. Inorganic Hg does not appear to biomagnify in aquatic food webs, however. There is no satisfactory explanation for the contradictory results of laboratory and field studies, with respect to Hg biomagnification.

#### Organic Compounds

##### Polychlorinated biphenyls

75. Polychlorinated biphenyls (PCB) in freshwater organisms. Laboratory studies indicate significant uptake of PCB in contaminated foods by freshwater animals. Lieb, Bills, and Sinhuber (1974) found that rainbow trout (*Salmo gairdneri*) fed a trout diet containing 15 ppm PCB (Aroclor 1254) retained 68 percent of the total PCB ingested over a 32-week feeding period (Table 36). Figure 3 shows that no depuration of accumulated PCB occurred when trout were removed from the PCB diet. Sommer et al. (1982) examined the uptake of PCB by yellow perch (*Perca flavescens*) fed control and experimental diets containing 0.2 and 1.8 ppm PCB, respectively. Table 37 shows that perch fed the control diet for 30 weeks contained about 25 percent as much PCB as those fed the diet containing 1.8 ppm of PCB. Biomagnification of PCB did not occur during the 30-week feeding period, but PCB accumulation was significant. The authors also indicated that adult fish depurated PCB less rapidly than fingerlings.

76. Bruggeman et al. (1981) also reported significant PCB uptake from food by goldfish (*Carassius auratus*). Table 38 shows that there was much greater accumulation of PCB from PCB-saturated water than from dietary exposure. This would suggest that fish acquire the majority of their PCB body burden by bioconcentration from the water, rather than from dietary uptake. PCB-saturated water probably does not occur in nature, however. As the PCB levels in the aqueous exposure study (>130 ppb) are very likely unrealistic,



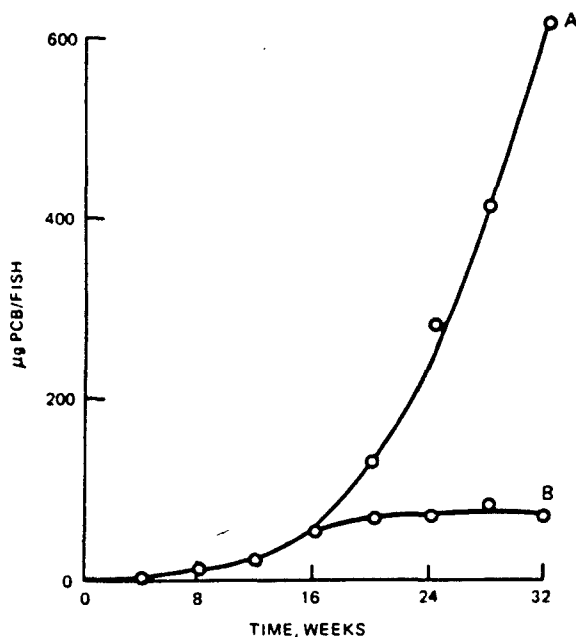


Figure 3. Total amount of PCB accumulated by fish from their diet: (A) fish on diet containing 15 ppm of PCB; (B) fish removed from diet containing 15 ppm of PCB at end of 16 weeks (from Lieb, Bills, and Sinnhuber 1974). Reprinted with permission from the Journal of Agricultural and Food Chemistry, Vol 22, p 640, Copyright 1974--American Chemical Society

the high levels of PCB in the goldfish may not necessarily reflect what occurs in nature. The National Research Council (1979) reported PCBs in the range of a few parts per trillion (ppt). That the PCB levels in water in the study by Bruggeman et al. (1981) were unrealistic has been shown clearly by Spigarelli, Thommes, and Prepejchal (1983), who reported that the PCB levels in filtered water from Lake Michigan were  $10.9 \pm 1.3$  ng/l (ppt). Adult brown trout (*Salmo trutta*) were exposed to lake water containing 10.9 ppt PCB  $\pm$  alewife (*Alosa pseudoharengus*) that were taken from Lake Michigan and which contained 2.5 ppm PCB. Figure 4 shows the accumulation of PCB from water alone and from food and water under different conditions of cyclic and constant temperature. The authors reported that uptake from the water alone accounted for about 10 percent of the total uptake from food and water. A cyclic temperature regime, more closely resembling a natural condition than constant temperature, gave substantially higher PCB uptake than constant 13° C.

77. Field studies have not given a clear picture of the potential for biomagnification of PCB in freshwater food webs. Veith (1975) presented evidence that biomagnification of PCB occurred in Lake Michigan fishes. Table 39 shows substantially greater PCB concentrations in the top predators, salmon and lake trout, than in fishes at lower trophic levels. In three Finnish lakes, however, Paasivirta et al. (1983) found greater PCB concentrations in

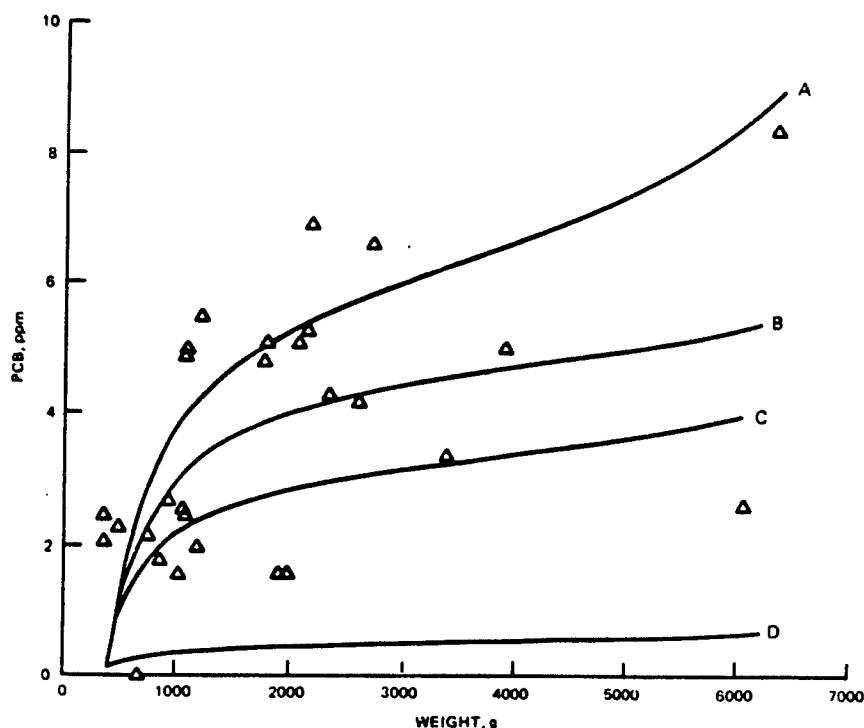


Figure 4. Projections of PCB concentrations (parts per million) in brown trout stocked into Lake Michigan at Age II (400 g) under four assumed conditions: (A) continuous exposure to cyclic temperature regime, PCBs in food = 2.5 ppm; (B) continuous exposure to ambient temperatures, PCBs in food = 2.5 ppm; (C) continuous exposure to constant 13°C, PCBs in food = 2.5 ppm; (D) continuous exposure to ambient temperatures, PCBs in food = 0. Data ( $\Delta$ ) are PCB concentrations in brown trout from Lake Michigan in 1974-1975 (from Spigarelli, Thommes, and Prepejchal 1983). Reprinted with permission from Environmental Science and Technology, Vol 17, p 93, Copyright 1983--American Chemical Society

the plankton than in fish (Table 40). The PCB levels in 2 top carnivore (pike) were not significantly different than those in roach. The PCB levels in fats were substantially higher in pike than in roach and higher in the fishes than in the plankton. Tsui and McCart (1981) similarly found no trend toward PCB biomagnification in fishes from Cold Lake, Alberta, if PCB concentrations in muscle were examined. Pike contained a higher mean concentration of PCB in fats than the other species, however.

78. PCB in marine organisms. Laboratory tests with marine organisms have produced some conflicting evidence regarding the route of PCB uptake.

Narbonne (1979) reported that the accumulation of Phenoclor DP6 in the liver, muscle, and carcass of grey mullet exposed for 48 hr to 0.5 ppm in water were 7, 66, and 16 times greater, respectively, than the accumulation over a 30-day period from food containing 50 ppm. Although the author concluded that food was not a major source of PCB for the mullet, uptake from food was indicated. The use of dimethylsulfoxide (DMSO) as a carrier to maintain the PCB in solution as well as unrealistic levels of PCB may have been responsible for the high levels of PCB accumulated by the mullet, as suggested recently by Spigarelli, Thommes, and Prepejchal (1983). Another study (Scura and Theilacker 1977) examined a three-step "food chain" exposed globally for 45 days to 2.3 to 2.1 ppb Aroclor 1254 in seawater without using a carrier solvent. At the end of the study, the concentrations of the PCB in algae (*Dunaliella* sp.), rotifers (*Brachionus plicatilis*), and larval anchovies (*Engraulis mordax*) were 0.25, 0.42, and 2.06 ppm, respectively. The authors state "what appeared to be bioamplification up the food chain when compared on a dry-weight basis was in reality only a reflection of the higher percentage of lipids in the rotifers." The authors' statement against biomagnification is untenable, however, as the average lipid contents of algae, rotifers, and anchovy larvae were 6.4, 15.0, and 7.5 percent, respectively. Anchovy larvae had only half as much lipid but five times the level of PCB as rotifers. Although these data show that PCB was being biomagnified, other data presented suggest that most of the PCB in anchovy larvae came from equilibrium partitioning between PCB in the water and PCB in lipids, rather than from food. Over a 25-day period, anchovy larvae feeding on rotifers averaged 2.06 ppm PCB in comparison with 2.80 and 4.70 ppm, respectively, in unfed 3- and 2-day-old anchovy larvae. Courtney and Langston (1980) reported the uptake of Aroclor 1254 by juvenile turbot (*Scophthalmus maximus*) exposed to the PCB via different sources. Table 41 shows that uptake from food occurred but was substantially less than the uptake from sediments. This suggests that uptake of PCB from sediments may be more important than that from food, at least in areas with high PCB contamination. If this also occurs in the field, the open-water disposal of large amounts of dredged materials that are highly contaminated with PCBs potentially may cause an undesirable impact upon aquatic organisms near the disposal areas. Such open-water disposal sites very likely would be capped to prevent environmental perturbations.

79. Other laboratory data indicate that food is an important source of

PCB. Zitko (1974) found significant uptake of Aroclor 1254 by juvenile Atlantic salmon fed on diets containing 10 and 100 ppm of the PCB. At the lower exposure level, no further uptake occurred after about 30 days feeding, indicating that equilibrium had been reached (Table 42). At 100 ppm PCB in the diet, however, equilibrium had not been reached after 181 days. Recent work by Rubinstein (N. I. Rubinstein, U.S. Environmental Protection Agency, Gulf Breeze, Florida, unpublished data) has demonstrated food-chain transfer of PCB from polychaete worms (*Nereis virens*) to spot (*Leiostomus xanthurus*). Spot fed for 20 days on worms containing an average of 0.45 ppm PCB contained about 1.05 ppm PCB in contrast to about 0.1 ppm PCB in fish fed control ("clean") worms. These results suggest that diet contributed significantly ( $p \geq 0.001$ ) to the PCB body burden in spot.

80. Some evidence has been shown that PCB may not biomagnify in natural marine food webs. Giam et al. (1972) found no tendency for PCB levels to increase with trophic level. Large predatory fishes contained PCB concentrations in muscle tissues that were less than or equal to those in smaller fishes and invertebrates (Table 43). The organisms in this study were collected on different dates and at different locations throughout the Caribbean Sea and Gulf of Mexico. Consequently, the comparison may not be entirely valid. Table 44 (Warfe and van den Broek 1978) shows substantially higher PCB levels in mussels, crabs, and shrimps than in whiting from the Lower Medway Estuary, Kent. Bastürk et al. (1980) reported similarly that fish (mullet and goatfish) had levels of PCB less than or equal to those in invertebrates (Table 45). The data were inadequate to make any valid conclusions, however. Schneider (1982) also showed no definite relationship of PCB level in marine organisms to trophic level, either on the basis of parts per million wet weight or parts per million in lipid (Table 46).

81. Using Cs:K ratios, Young and Mearns (1979) found poor correlations between trophic level and total PCB concentrations in two of three marine food webs. In the Palos Verdes food web, fishes had somewhat greater PCB burdens than invertebrates, with the exception of bocaccio, which was at the top of the trophic web, but had relatively low PCB levels. Similar work by Schafer et al. (1982) showed widely varying PCB levels at all trophic levels in three California marine food webs. The highest levels observed in each food web were in predatory species, however (Table 47). Unusually high levels, 41.9 and 84.7 ppm PCB, were reported for spiny dogfish and sea lions, respectively.

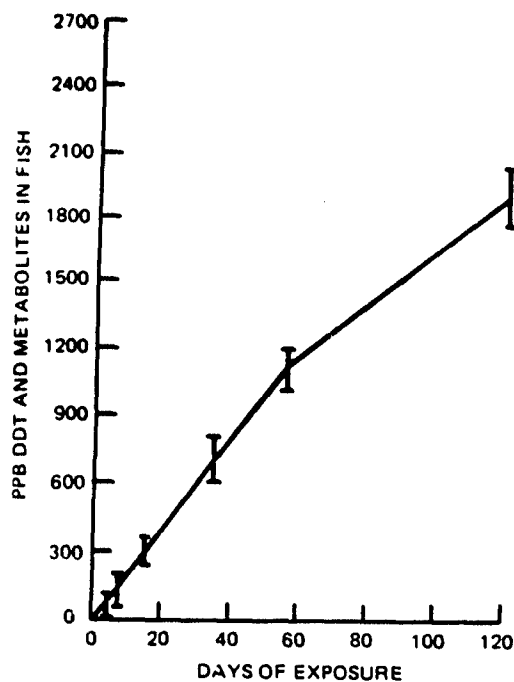
Overall, predators feeding primarily on fish had somewhat to significantly higher PCB levels than organisms at the bottom levels of the trophic webs.

82. Other field studies indicate a definite trend for PCB amplification at the higher levels of the food web. Jensen et al. (1969) reported significantly greater PCB concentrations in the top predators, than in other marine organisms collected off the Swedish coast. Mowrer et al. (1977) reported higher levels in cottid fish (sea robins) than in mussel in Puget Sound (Table 48). Similarly, Goerke et al. (1979) found significantly higher PCB levels in sole (*Solea solea*) than in invertebrates from the Weser Estuary (Table 49). PCB concentrations in fish samples from Ora, Norway, were significantly greater (Table 50), both on a wet weight and fat basis, than those in most invertebrates (Bjerk and Brevik 1980). Crabs (*Carcinus maenus*) had PCB levels that were not significantly different than those of gobies, however. Courtney and Langston (1980) reported higher PCB concentrations in flounder than in cockle (Table 51). The trend toward biomagnification of PCB was not clear in the Brisbane River Estuary, Australia (Shaw and Connell 1980), where fishes and invertebrates had similar PCB concentrations (Table 52).

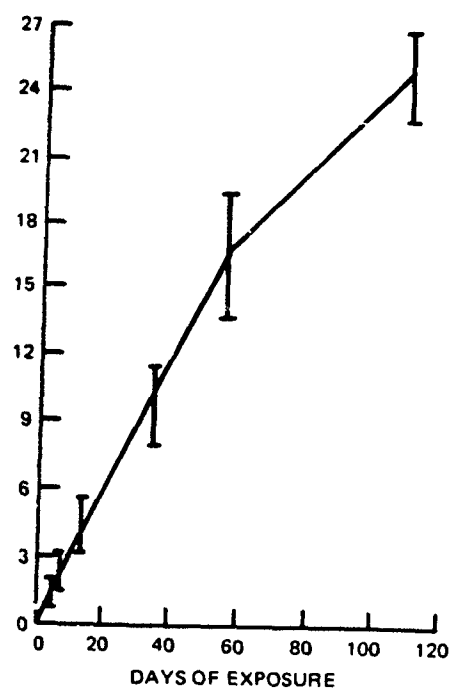
#### DDT and its derivatives

83. DDT in freshwater organisms. The significance of the food chain in bioaccumulation of DDT has been investigated in several laboratory studies. Macek and Korn (1970) found that brook trout (*Salvelinus fontinalis*) accumulated significantly more DDT from food than from water. At DDT levels in food (diet pellets containing 3 ppm DDT) and water (3 ppb) similar to those observed in nature, total uptake of DDT from food over a 120-day exposure period was about two orders of magnitude greater than that from the water (Figure 5). Fish exposed separately to DDT in the water and via food accumulated 3.6 and 35.5 percent, respectively, of the total available DDT. Since the DDT level in food is generally greater than in the water, the authors concluded that food is the major source of DDT for aquatic organisms. Hamelink, Waybrant, and Ball (1971) reported that the food chain was less important in DDT bioaccumulation than direct exchange of DDT between the water and fats, based upon differential solubility. The food-chain study was based upon a global contamination design including algae, invertebrates, and fish. The data presented demonstrate biomagnification of DDT, regardless of which route of entry (food or water) was more important (Table 53).

84. Field studies that have examined DDT levels in freshwater organisms



a. 3 ppm p,p'-DDT in the diet



b. 3 ppb p,p'-DDT in the water

Figure 5. Total residues (DDT, DDD, DDE) accumulated by brook trout exposed for 120 days (from Macek and Korn 1970). Reprinted with permission from the Journal of the Fisheries Research Board of Canada, Vol 27, p 1497, Copyright 1970--Fisheries and Oceans Scientific Information and Publication Branch

leave a confusing picture of the behavior of DDT within the freshwater food web. Data from Tule Lake Wildlife Refuge (Table 54) show generally higher levels of DDT in fish (chubs) than in other biotic components of the ecosystem (Godsil and Johnson 1968). Widely varying levels of DDT with sampling date and inconsistent sampling of the various biotic components over time preclude any valid conclusion that DDT biomagnified within this ecosystem. Paasivirta et al. (1983) reported that mean DDE concentrations in Finnish lakes were highest in plankton and decreased progressively from roach to pike, apparently the top predators in these lakes. Table 40 shows that DDE levels varied widely and probably were not statistically different in the three components of the lake ecosystem.

85. Mack et al. (1964) reported that DDT residues in fish from various lakes in New York were generally highest in the top predators and lower in other species. Table 55 shows this trend for DDT in fishes at the Taughannock

location in Cayuga Lake (the only location with a sufficient number of species collected simultaneously to constitute a legitimate trophic chain). Veith (1975) similarly reported higher average concentrations of DDE and total DDT in the top predators (salmon and lake trout) in Lake Michigan, suggesting possible biomagnification of DDT and its derivatives within fishes in this system (Table 39). Comparatively large standard deviations suggested either that DDT in different fish species varied widely with sampling location and/or date or that there were very few significant differences in DDT levels across the entire food chain. Tsui and McCart (1981) found no evidence for the biomagnification of DDT in fishes from Cold Lake, Alberta. Table 56 shows that total DDT concentrations in muscle tissues of pike were similar to those in cisco, whitefish, and suckers. When average DDT levels were reported on a fat basis, however, DDT appeared to biomagnify, as DDT and its derivatives concentrate in the fat fraction. Concluding biomagnification of DDT on the basis of concentrations in fats may be misleading, as can be shown by the data of Bulkley, Leung, and Richard (1981). Table 57 shows the percent body fat and whole-body levels of DDT in seven species of fish from the Des Moines River. The highest body burdens of DDT were in the larger predators (walleye, catfish, and bass), all of which contained relatively low amounts of body fat, in contrast to 17 percent fat content in shad. Lincer et al. (1981) found no evidence that would indicate biomagnification of DDE in aquatic organisms in three Kenya lake drainage systems (Table 58).

86. DDT in marine organisms. Some of the evidence from field sampling suggests that DDT may biomagnify within marine food webs. Goerke et al. (1979) found generally higher levels of DDE and DDD in common sole (*Solea solea*) in the Weser Estuary of the North Sea. There was no obvious pattern that would suggest a relationship between DDD or DDE levels and invertebrate trophic levels, however (Table 49). Courtney and Langston (1980) reported substantially higher levels of DDT and its metabolites in flounder than in cockles (Table 51). Basturk et al. (1980) found similarly that the level of DDT and DDE residues was generally greater in fishes than in the invertebrates. Table 45 shows that, although the ranges overlapped, mean and maximum DDT residues were significantly higher in two species of mullet and in goatfish than in either shrimp or limpets.

87. Studies using Cs:K ratios to assign trophic levels indicate that DDT levels may be higher at the upper trophic levels than at lower ones, but

the pattern is not entirely clear. Young and Mearns (1979) found substantially higher DDT levels in scorpionfish and sanddab than in molluscs and crustaceans in the Palos Verdes food web (Table 59). Yellow crab, however, had higher DDT levels than did bocaccio, supposedly at the top trophic level. In the Salton Sea and Newport Bay food webs, however, no such relationship existed. The correlation between percent lipids and total DDT was good in the Newport Bay food web. If mullet, which had both the highest lipid content and highest DDT levels, were not included, there would be a definite trend toward biomagnification of DDT at Newport Bay. Lipid and DDT content were poorly correlated in the other two food webs, however. A subsequent study (Schafer et al. 1982) showed generally higher DDT levels in top predatory fishes than in invertebrates (Table 47). The high DDT concentrations (24.8 ppm) found in sea lions in the California coastal pelagic food web probably reflect their high lipid content and the relatively impervious integument required for a partially terrestrial existence. The low DDT levels in basking sharks and blue whales very likely reflect both their diet (plankton) and their habit of living in deep waters, often far from the more contaminated environments.

88. Other field studies indicate that biomagnification of DDT does not occur to any great extent within most marine invertebrate and fish food webs. In the Gulf of Mexico and the Caribbean Sea, DDT levels (Table 43) in the muscle tissue of large fishes generally were lower than those in invertebrates and those in smaller fishes (Giam et al. 1972). Levels in livers usually exceeded those in the muscles. A comparison of DDT levels (Table 44) in mussels, crabs, shrimps, and fish from the Lower Medway Estuary, Kent, also showed no trend toward biomagnification (Wharfe and van den Broek 1978). The level of DDT in fish livers exceeded that in muscle tissue by about two orders of magnitude. Bjerk and Brevik (1980) showed similarly that DDT levels in fish were not significantly greater than those in invertebrates upon which they feed (Table 13). Levels of DDT in fats were generally about two orders of magnitude greater than those in whole tissues. Biomagnification did not occur either on the basis of DDE in fats or DDE in whole tissues. Robinson et al. (1967) reported DDT levels ranging from 0.003 to 0.16 ppm in invertebrates and 0.012 to 0.080 ppm in fish (Table 60). Woodwell et al. (1967) ranked organisms in the Carmans River Estuary, New York, according to increasing DDT concentrations. The highest DDT concentrations in water-breathing animals were found in three species of fishes (mummichog, flounder, and chain



pickerel), with lower concentrations in the other fauna (Table 61). Jensen et al. (1969) found that DDT levels generally increased from mussels to fish in marine species along the Swedish coast (Table 62).

#### Other organochlorine pesticides

89. Dieldrin and endrin. The available data on these insecticides show a somewhat confusing, if not contradictory, picture of the behavior of these chemically similar compounds within aquatic food webs. Reinert (1972) compared the uptake of dieldrin from water and food in an alga-*Daphnia*-guppy food chain. *Daphnia magna* exposed to dieldrin in water containing algae (*Scenedesinus obliquus*) accumulated slightly less dieldrin than from exposure to dieldrin in water alone. The dieldrin levels in water and algae, respectively, were 2.2 to 4.4 ppb and 4.2 to 7.5 ppm. Guppies fed equal daily rations of *Daphnia* containing different dieldrin concentrations accumulated dieldrin concentrations that were directly proportional to the concentrations in the *Daphnia*. Figure 6 shows that the major source of dieldrin for guppies was water, however. In another laboratory study, Petrocelli, Anderson, and Hanks (1975) demonstrated the uptake of dieldrin by blue crabs (*Callinectes sapidus*) fed for 5 to 10 days on clams (*Rangia cuneata*). Table 63 shows that the uptake of dieldrin from food was significant. The authors stated that biomagnification of dieldrin was possible.

90. Field-collected data also present an unclear picture of biomagnification of these compounds. Robinson (1968) presented data on dieldrin levels in organisms at various trophic levels within an unspecified marine ecosystem. Figure 7 shows that the average dieldrin levels were similar within the various trophic levels of the obligate aquatic fauna. Another study by Robinson et al. (1967) reports dieldrin levels in various marine organisms from off the Northumberland coast. Dieldrin levels increased from algae to microzooplankton and then fluctuated widely throughout the higher trophic levels (Table 60). The dieldrin levels in fishes varied from 0.002 to 0.038 ppm. Warfe and van den Broeck (1978) showed high levels of dieldrin in molluscs (*Mytilus edulis*) with lower levels in crustaceans and fish. Table 44 shows that, on the basis of dieldrin in muscle, dieldrin does not biomagnify through the trophic web. Very high levels were present in the livers of the fish, however.

91. Other studies have reported dieldrin concentrations normalized on a lipid basis. Goerke et al. (1979) reported a similar pattern for dieldrin in

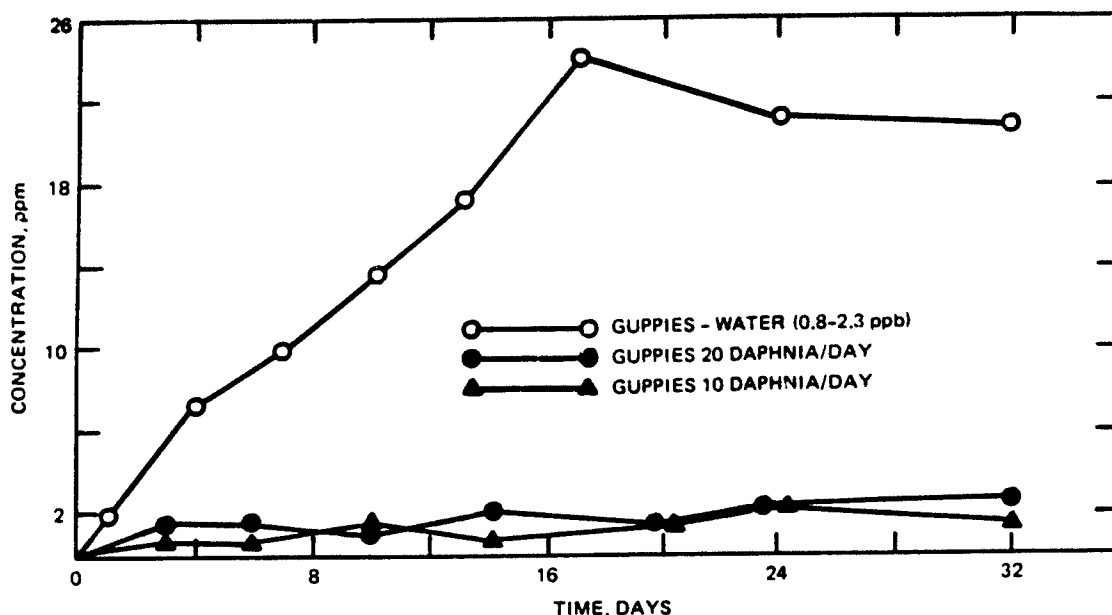


Figure 6. Dieldrin concentrations in guppies held in water containing 0.8-2.3 ppb dieldrin and in guppies held in dieldrin-free water but fed two quantities of *D. magna* containing an average of 32 ppm dieldrin. Each point represents the average for two fish. These *D. magna* were raised in water with a dieldrin level of 1.8-2.5 ppb (from Reinert 1972). Reprinted with permission from the Journal of the Fisheries Research Board of Canada, Vol 29, p 1417, Copyright 1972--Fisheries and Oceans Scientific Information and Publications Branch

animals from the Weser Estuary (Table 49). On either a whole-tissue or lipid basis, dieldrin did not biomagnify within the food chain. Dieldrin concentrations were about two orders of magnitude greater when expressed on a lipid basis rather than on whole-tissue concentrations. Pick, de Beer, and van Dyk (1981) investigated dieldrin residues in fats of several species of fish from the Transvaal, South Africa. Data from the same locations indicate that dieldrin residues probably do not biomagnify in fish (Table 64). Bulkley, Leung, and Richard (1981) show data for fishes in the Des Moines River, Iowa, that suggest that normalization on a lipid basis is not meaningful for dieldrin (Table 57). Fishes containing a high percentage of body fat did not contain correspondingly high levels of dieldrin residues. Dieldrin was not biomagnified in this study.

92. Endrin residues in aquatic biota of the Tule Lake Wildlife Refuge varied with time but showed a possible trend toward biomagnification (Godsil and Johnson 1968). The highest endrin concentration (198 ppm) was observed in chubs collected on 27 August 1965, but no other data were available for that

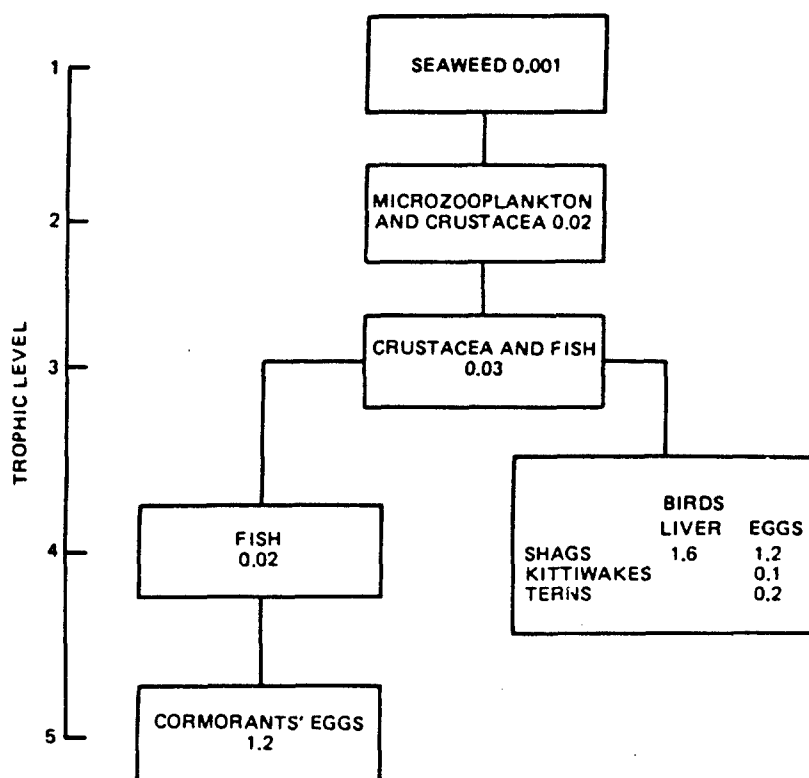


Figure 7. Concentrations of HEOD (parts per million) in the organisms of various trophic levels of a marine ecosystem (from Robinson 1968). Reprinted with permission from Chemistry in Britain, Vol 4, p 160, Copyright 1968--The Royal Society of Chemistry

date (Table 54). Chubs contained generally higher levels of endrin than other organisms collected on the same dates. The data were insufficient to positively conclude biomagnification of endrin, however.

93. Lindane (BHC). Both laboratory and field studies indicate that lindane (BHC) is not accumulated to any extent from food and, consequently, does not biomagnify with aquatic food webs. Streit (1979) reported that diatoms (*Nitzschia actinastroides*) grown in 10 ppb lindane absorbed lindane, but the lindane was not transferred to any degree to freshwater limpets (*Ancylus fluviatilis*). Figure 8 shows the concentration factor in limpets feeding and not feeding on the algae. Hansen (1980) demonstrated rapid lindane uptake from water by algae (*Chlorella* sp.), *Daphnia* sp., and the stickleback (*Gasterosteus aculeatus*). Figures 9 and 10 show that the uptake of lindane (10 ppb) from water was substantially greater than that from food for *Daphnia* feeding on algae and sticklebacks feeding on *Daphnia*, respectively.

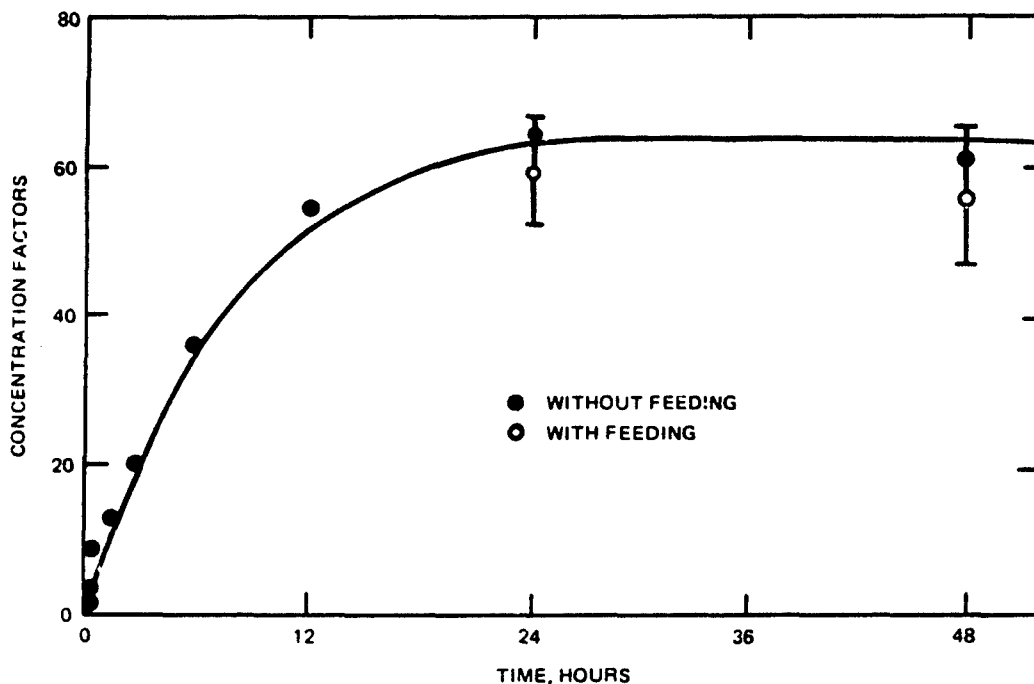


Figure 8. Concentration factors (CF) of lindane on a fresh weight basis by *Ancyclus fluviatilis*, fed ad libidum on contaminated algae. During a 2-day period, feeding (open circles) and non-feeding limpets (solid dots) were compared. Their CF values were not significantly different. Values of feeding specimens are represented by arithmetic means and standard deviations (from Streit 1979). Reprinted with permission from *Archiv für Hydrobiologie, Suppl.*, Vol 55, p 387, Copyright 1979--E. Schweizerbart'sche Verlagsbuchhandlung

Field studies by Tsui and McCart (1981) indicate no trend toward biomagnification of BHC in the fishes of Cold Lake, Alberta. Table 56 shows similar concentrations of lindane in pike as in other fishes. Likewise, Pick, de Beer, and van Dyk (1981) noted similar levels of BHC in fishes from the Transvaal of South Africa (Table 64).

94. Kepone and mirex. Laboratory studies indicate that these two persistent, structurally similar insecticides may be absorbed either through the water or from contaminated food. Bahner et al. (1977) examined kepone transfer in algae-oyster and plankton-mysid-fish food chains. In both food chains, kepone was transferred up the food chain via food. The average kepone residues in algae following 24 hr of exposure to kepone and in oysters fed for 14 days on contaminated algae were 34 and 0.21 ppm, respectively. In the plankton-mysid-fish food chain, however, substantial quantities of kepone were

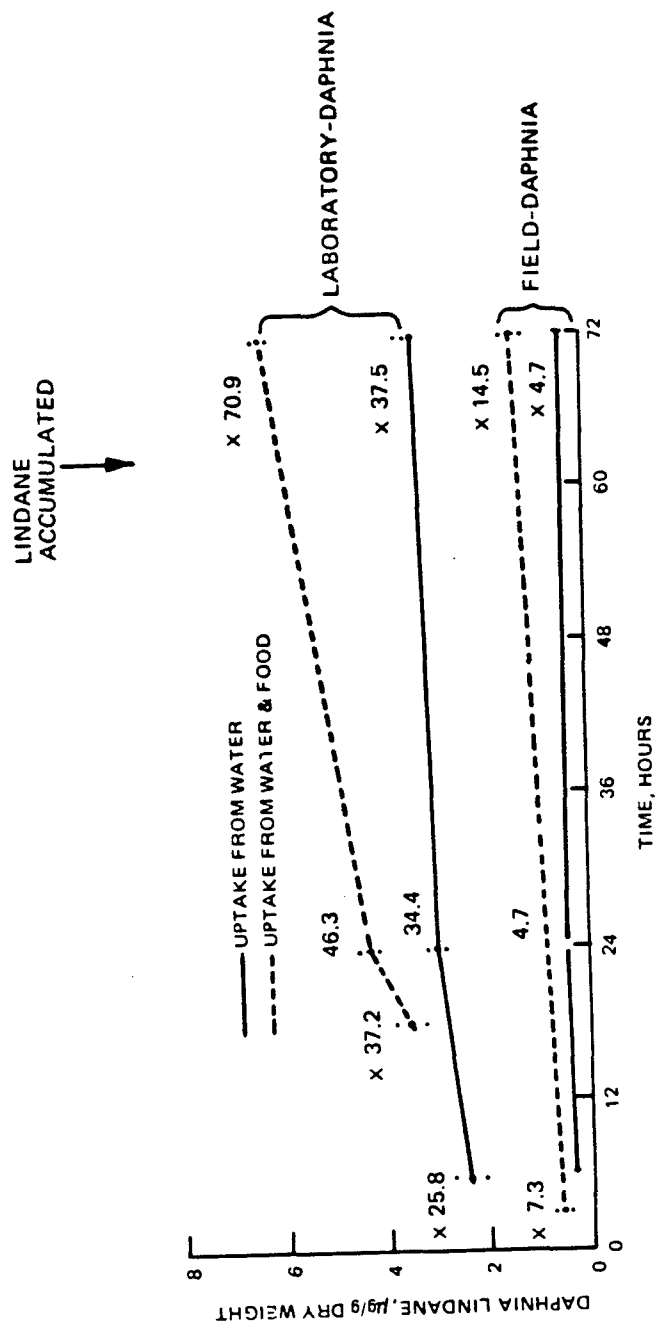


Figure 9. Uptake and accumulation factors for lindane by *Daphnia magna* from the water at 10 µg lindane per litre, and from food (from Hansen 1980). Reprinted with permission from *Environmental Pollution*, Vol 21, p 101, Copyright 1980--Applied Science Publishers

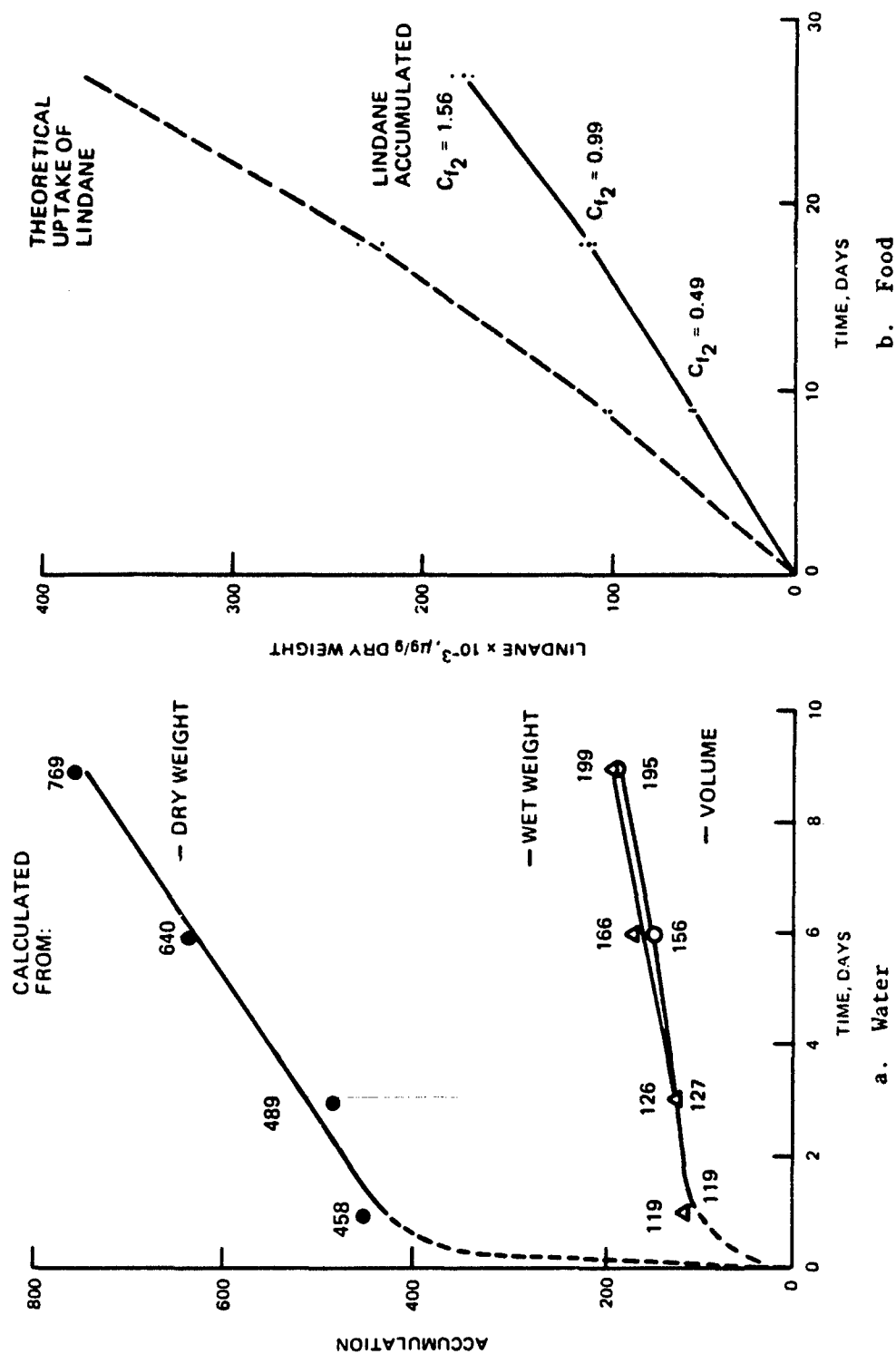


Figure 10. Accumulation of lindane from water at 10  $\mu\text{g}/\ell$  and uptake of lindane via food by *Gasterosteus aculeatus* in water free from lindane.  $c_{f2}$  = accumulation factor of the fish (Consumer 2) by uptake via food (from Hansen 1980). Reprinted with permission from *Environmental Pollution*, Vol 21, p 103, Copyright 1980--  
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passed from the plankton (brine shrimp) through the mysids to the fish (spot). Table 65 shows that mysids feeding on plankton accumulated about half the level of kepone in the food. Spot fed for 30 days on contaminated mysids accumulated kepone to levels about 85 percent of that in the food. This work was continued by Schimmel et al. (1979), who reported the uptake of kepone by blue crabs (*Callinectes sapidus*) fed for 28 days on oysters contaminated with 0.25 ppm kepone. Figure 11 shows that kepone uptake was similar whether via food alone, food and 0.03 ppb kepone in the water, or food and 0.3 ppb kepone in the water. No significant loss of kepone occurred during a 28-day depuration period. A model for predicting kepone accumulation has been developed

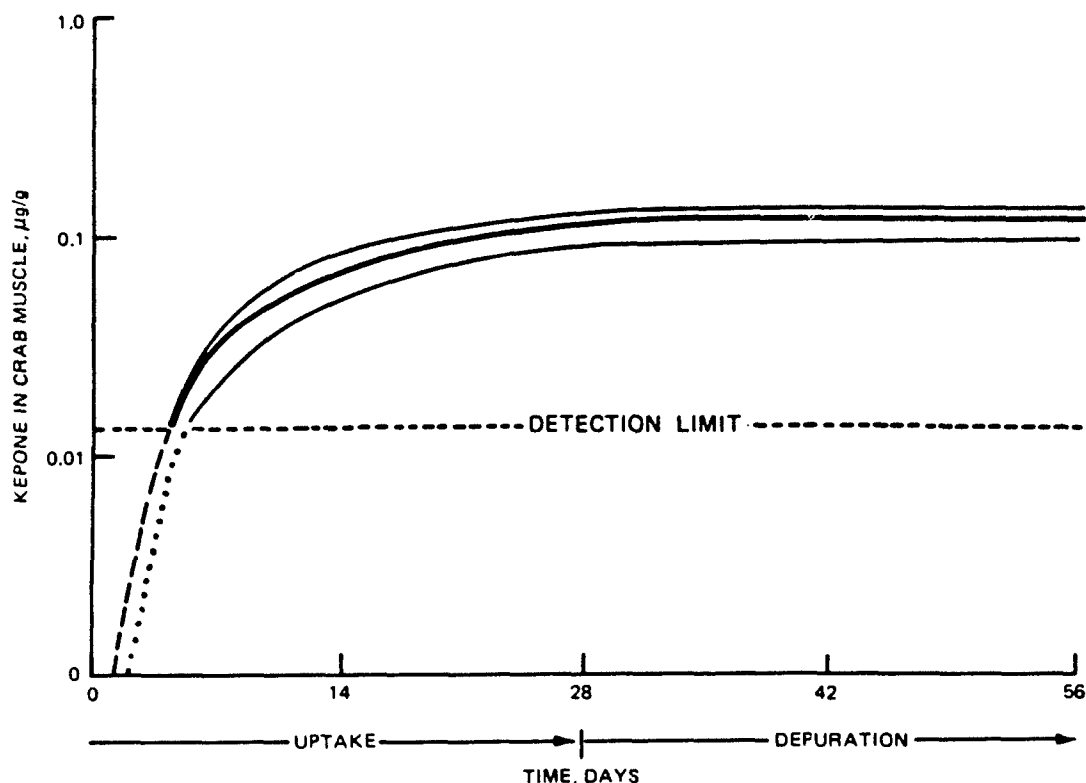


Figure 11. Bioaccumulation of kepone in muscle tissues of blue crabs (*Callinectes* spp.) fed oysters contaminated with 0.25 µg/g of the insecticide for 28 days followed by a 28-day period of depuration. The uptake curve (dark line) and the 95-percent confidence interval (lighter lines) represented are a composite of three homogeneous curves representing uptake in crabs fed: (1) 0.25 µg/g kepone in oysters and control seawater, (2) 0.25 µg/g in oysters and 0.03 µg/l in seawater, and (3) 0.25 µg/g in oysters and 0.3 µg/l in seawater (from Schimmel et al. 1979). Reprinted with permission from Estuaries, Vol 2, p 12, Copyright 1979--Estuarine Research Foundation

from these studies by Bahner and Oglesby (1979). Studies by Skaar et al. (1981) confirmed the uptake of kepone and mirex by bluegills from food (*Daphnia*). Figure 12 shows significantly greater uptake of mirex than kepone by bluegill feeding on contaminated *Daphnia*. This difference appeared to be the result of significant depuration of kepone competing with accumulation. Mirex levels in bluegill did not decrease over a 28-day depuration period.

95. Atrazine. The herbicide atrazine has not been shown to accumulate extensively in aquatic food-chain organisms. Streit (1979) indicated that little atrazine was transferred via feeding on atrazine-contaminated algae. The concentration factors for atrazine in limpets were similar regardless of whether or not the limpets were feeding (Figure 13). McEnerney and Davis (1979) demonstrated trophic transfer of  $^{14}\text{C}$ -labelled atrazine from *Spartina*

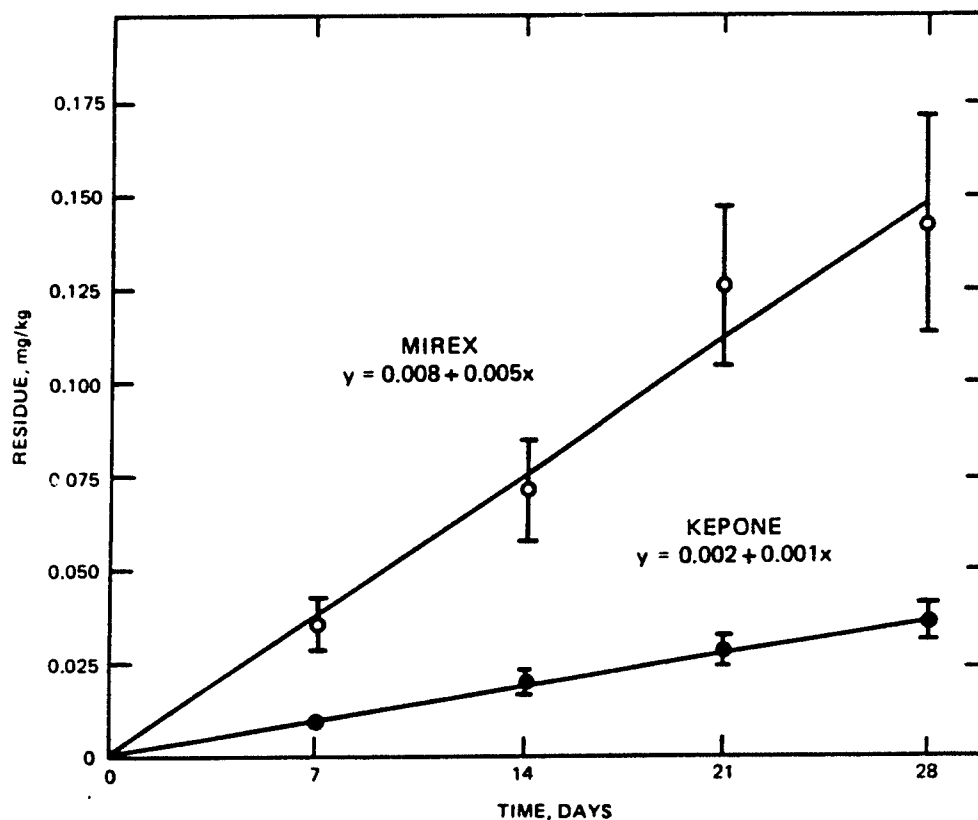


Figure 12. Mean accumulation of kepone and mirex by bluegills from food (*Daphnia*) over a 28-day-period. Vertical bars show standard deviation (from Skaar et al. 1981). Reprinted with permission from the Canadian Journal of Fisheries and Aquatic Sciences, Vol 38, p 935, Copyright 1981--Fisheries and Oceans Scientific Information and Publications Branch



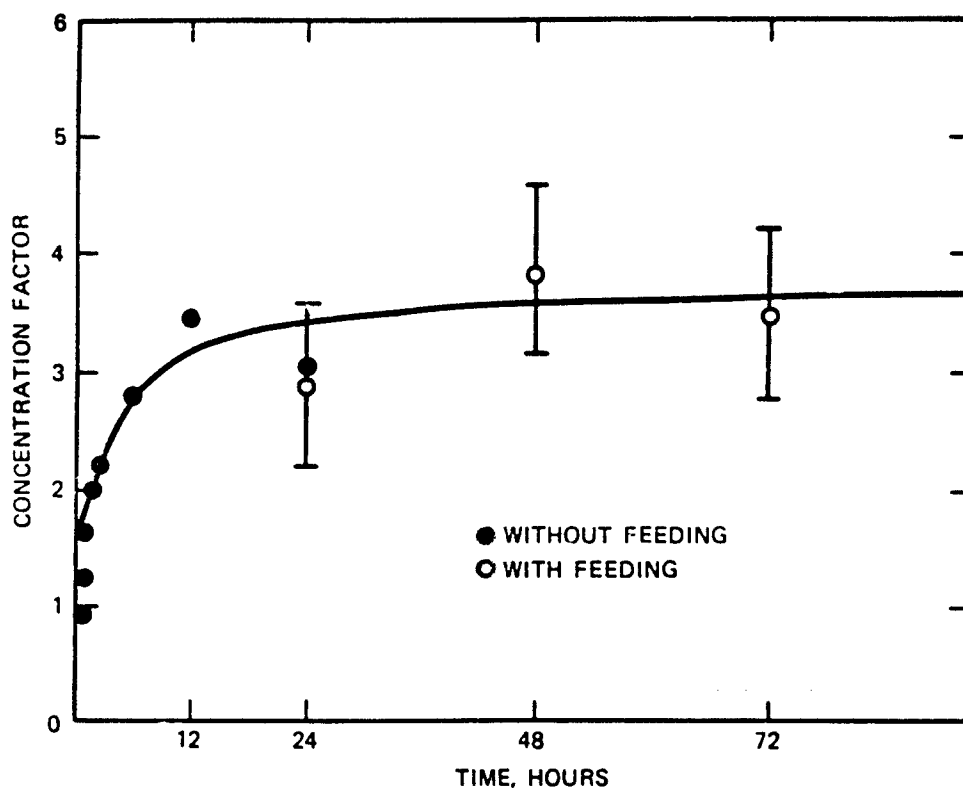


Figure 13. Concentration factor (CF) of atrazine in limpets on a fresh weight basis without feeding (solid dots) and when feeding (open circles: arithmetic mean and standard deviation of single values) (from Streit 1979). Reprinted with permission from Archiv für Hydrobiologie, Suppl., Vol 55, p 384, Copyright 1979-- E. Schweizerbart'sche Verlagsbuchhandlung

*alterniflora* detritus to fiddler crabs (*Uca pugnax*). The authors indicated that atrazine and atrazine metabolite levels decreased from detritus to crabs.

96. Endosulfan. Very little information is available regarding the uptake of endosulfan. The data available suggest that endosulfan does not bio-magnify in marine organisms. Goerke et al. (1979) reported a slightly higher mean level of endosulfan in sole than in invertebrates, but the level in sole was not significantly different than in the clam, *Mya arenaria* (Table 49). Pick, de Beer, and van Dyk (1981) reported 5.88 ppm endosulfan in yellowfish, 1.09 ppm in barbel, and 0.17 ppm in kurper (Table 64).

#### Miscellaneous organochlorine compounds

97. Chlorinated phenols. Very little aquatic food-chain information is available for the chlorinated phenolic compounds. Paasivirta et al. (1980) reported the levels of several chlorinated phenols in the biota of lakes in

Finland. Table 66 shows that, of the six chlorophenols examined, none of the compounds biomagnified. The mean levels of the different compounds frequently were higher in pike or roach than in mussels, sponges, or plankton. Wide standard deviations (and coefficients of variation) indicate that chlorophenol concentrations in the top trophic levels are not significantly different than those at lower levels.

98. Chlorinated benzenes. Evidence available suggests that the chlorinated benzene compounds probably do not biomagnify in aquatic systems. Macek, Petrocelli, and Sleight (1979) exposed bluegill for 35 days to TCB (1,2,4-trichlorobenzene) via aqueous and dietary routes and concluded that the diet did not contribute significantly to the TCB body burden in bluegill (Table 67). Bjerk and Brevik (1980), however, showed somewhat higher levels of HCB (hexachlorobenzene) in flounder than in gobies and mussels (Table 50). The levels in flounder were not significantly different than those in crabs, however. Similarly, Tsui and McCart (1981) found no evidence of HCB biomagnification in fishes from Cold Lake, Alberta (Table 56). Paasivirta et al. (1983) found no significant increase of HCB with trophic level in lakes in Finland (Table 40).

#### Polynuclear aromatic hydrocarbons

99. Laboratory studies have indicated trophic transfer of polynuclear aromatic hydrocarbons (PAH) from food to consumer, but the question of whether or not biomagnification occurs within aquatic food webs remains unresolved. Dobroski and Epifanio (1980) found significant uptake (18.6 ppm) of benzo[a]pyrene (B[a]P) by larvae of the clam, *Mercentaria mercenaria*, fed for 9 days on algae (*Thalassiosira pseudonana*) containing 42.2 ppm of B[a]P. At 9 days uptake was still linear, suggesting that biomagnification of B[a]P might occur (Figure 14). Significant depuration occurred within 24 hr after feeding ceased, however. Similarly, Dillon (1982) demonstrated significant dietary accumulation of dimethylnaphthalene (DMN) in grass shrimp (*Palaemonetes pugio*) fed for 32 days on contaminated brine shrimp (*Artemia* sp.). Table 68 shows the levels of DMN in brine shrimp and grass shrimp following exposure and after a recovery (depuration) period. The data suggest that DMN might biomagnify in marine invertebrates. A field study (O'Connor, Klotz, and Kneip 1982) suggests that naphthalene and several other PAHs (phenanthrene, anthracene, biphenyl, and total PAHs) do not biomagnify within aquatic systems (Table 69).

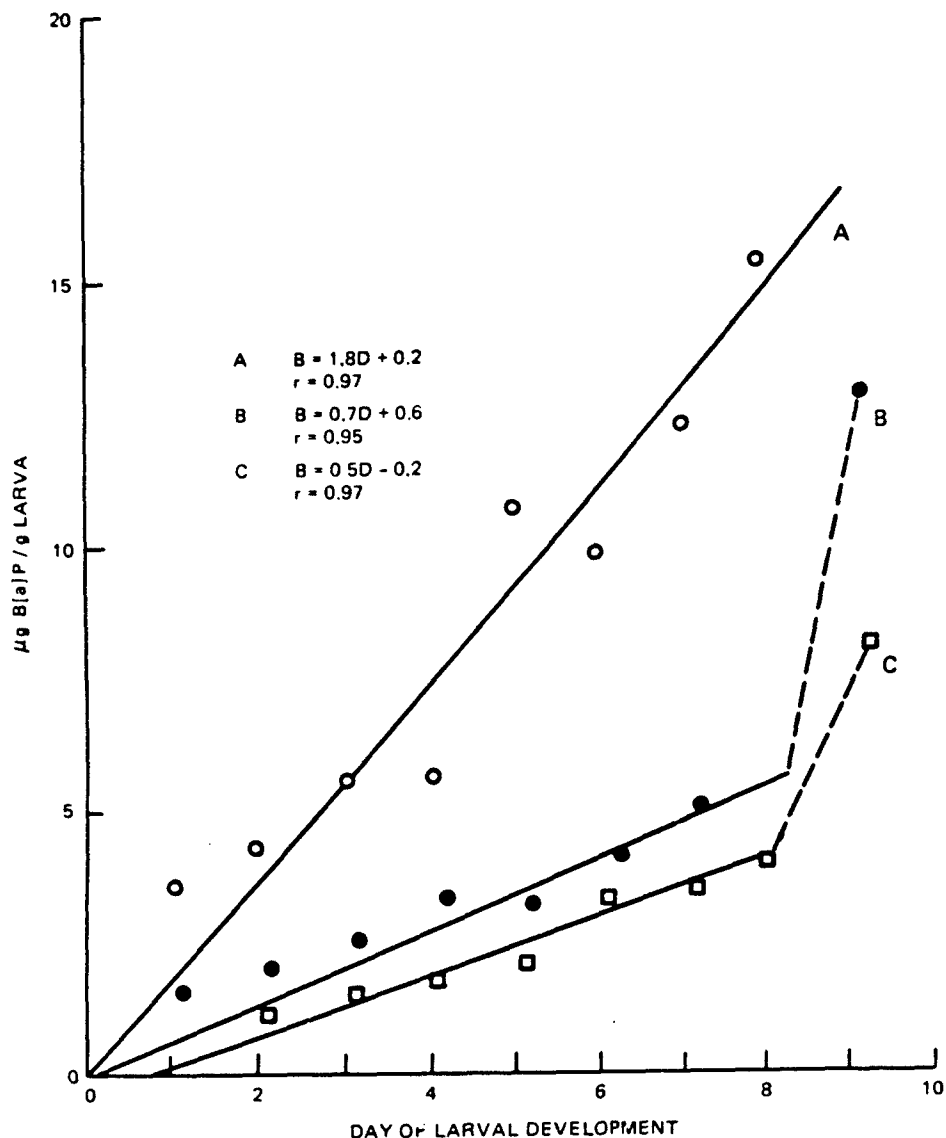


Figure 14. Accumulation and depuration of B[a]P by larvae of the clam *Mercenaria mercenaria*. Line A represents accumulation of  $^{14}\text{C}$ -benzo[a]pyrene by larvae fed contaminated algae; line B represents retention of  $^{14}\text{C}$ -benzo[a]pyrene by larvae after 24 hr depuration; line C represents retention of  $^{14}\text{C}$ -benzo[a]pyrene by larvae after 48 hr depuration (from Dobroski and Epifanio 1980). Reprinted with permission from the Canadian Journal of Fisheries and Aquatic Sciences, Vol 37, p 2321, Copyright 1980--Fisheries and Oceans Scientific Information and Publications Branch

### Summary

100. Food-chain studies with gill-breathing marine and freshwater animals indicate that food may contribute to the body burdens of a number of chlorinated and nonchlorinated organic compounds. Those which appear to have potential for biomagnification in aquatic food webs are the PCBs, kepone and mirex, benzo[a]pyrene, and naphthalenes. As in the case of the heavy metals, the data on these organic contaminants were frequently contradictory. Although top predatory fishes sometimes contained higher levels of specific contaminants than other members of the food web, the relationship between contaminant levels in the tissues and an organism's position in the food web was not clear. The apparent inconsistency in the data may reflect a number of factors including the mobility of the top predators, age and size differences, inadequate understanding of the feeding habits of different species (particularly with respect to the changing of feeding habits at different stages of the life cycle), imprecision in the assignment of trophic levels, and inadequate sampling and analytical procedures. Compounds which probably do not biomagnify include DDT (and its derivatives), lindane, atrazine, endosulfan, chlorinated phenols, chlorinated benzenes, and, probably, most of the PAHs, including phenanthrene, anthracene, and biphenyl. Relatively little information is available regarding the behavior of most of these compounds in aquatic food webs, however. Consequently, any absolute statement regarding biomagnification of these contaminants must be reserved until further data are available.

### PART III: CONCLUSIONS AND RECOMMENDATIONS

#### Conclusions

101. The information reviewed in this document indicates that biomagnification of contaminants is not a dramatic phenomenon in marine and freshwater food webs. Most heavy metals and organic compounds probably do not biomagnify over several trophic levels in aquatic ecosystems. Those contaminants which may have the potential to biomagnify include methylmercury, PCB, benzo[a]pyrene, naphthalenes, and possibly kepone and mirex. As the biological availability of contaminants from sediments should be similar regardless of whether or not these sediments have been dredged and placed in an open-water disposal site, it is unlikely that the open-water disposal of contaminated dredged material will cause any widespread ecological perturbations due to biomagnification.

102. Most of the evidence for the existence or non-existence of biomagnification within aquatic food webs has come from highly circumstantial and/or marginally relevant data. That which is known about the behavior of a very few specific contaminants (e.g., Cd, Pb, Cu, Hg, DDT, PCB, etc.) in aquatic ecosystems has been extrapolated to be all-inclusive of the general groups of compounds or elements to which these contaminants belong. The use of poor experimental design also has rendered a considerable body of data essentially useless. Many potentially good studies have been compromised either by poor data analysis or by unwillingness to reject a soundly defeated hypothesis. Consequently, before concluding that a given contaminant does or does not biomagnify, research is needed using carefully planned experimental designs that directly address the question at hand and produce results that are accurate and readily repeatable. Before concluding for or against biomagnification, laboratory data also must be compared with field data. On the basis of the literature review herein, several important ideas have emerged that will add credibility to the results of future research. These are summarized below.

#### Recommendations

103. The following are the reviewer's recommendations for improving the quality of laboratory and field-collected data related to the movement of contaminants in aquatic food webs.

### Laboratory studies

- a. For studies involving organic compounds, concentrate on those which have low water solubility and high fat solubility. Report the fat content of the organisms. Food-chain studies indicate that the tendency for compounds to biomagnify is inversely proportional to their water solubility (Hamelink et al. 1971; Geyer et al. 1982).
- b. Avoid the use of organic solvents, carriers, chelators, or any other compounds that enhance the solubility and/or availability of a contaminant to the target organism. Using such "solubilizers" may result in uptake characteristics that do not occur under natural conditions and lead to misinterpretation of the potential consequence of environmental exposure. Data from such studies are responsible for many of the contradictions seen in the literature on the relative importance of uptake from food versus direct absorption (bioconcentration).
- c. Use exposure levels and durations that are representative of those the organism would encounter in nature, either in its food or in its physical environment.
- d. Use organisms for food-chain studies that actually are representative of those in a natural ecosystem. If the top predator is a cold-water species (e.g., rainbow trout), do not feed it on a warm-water species (e.g., bluegill) or a tropical species (e.g., guppies).
- e. Remember that environmental conditions during exposure are important. Constant temperatures rarely occur in nature. Spigarelli, Thommes, and Prepejchal (1983) demonstrated clearly that uptake is affected by temperature regime (constant versus cyclic, as in nature). Organisms may respond to temperature, light, and tide (diurnal and diel cycles) and, consequently, behave differently in the laboratory than in nature. Laboratory conditions of exposure should resemble those in the organisms' natural habitat as closely as possible, so as not to disrupt its natural cycles. Collecting an organism in the late fall and subjecting it suddenly to long day lengths, bright light, and warm temperatures of summer conditions may have adverse effects on its physiology and life cycle and, due to metabolic perturbations, cause abnormal behavior under laboratory conditions during experimentation.
- f. Ensure that laboratory studies concerned with bioaccumulation also look at depuration of the contaminant. Use flow-through systems to maintain constant conditions during exposure or depuration.
- g. Consider carefully the effect of the chemical species (i.e., form) of a contaminant upon its bioaccumulation, partitioning within various tissues, and persistence, when designing the experiment.
- h. Consider the fact that the organism's requirement for an essential metal (e.g., Cu, Se, and Zn) may affect its ability

to accumulate and depurate that metal as well as other contaminants.

- i. Report radioisotope tracer data in terms of concentration (ppm, ppb, etc.) as well as in terms of specific activity whenever using radioisotope tracers to demonstrate the movement of contaminants in artificial food chains. Also report the ratio of radioactive to non-radioactive contaminant in each link of the food chain, both before and after introducing the radioisotope tracer.
- j. Determine initial background levels of contaminants in experimental organisms. Without this information, uptake studies are meaningless.

#### Field studies

- a. Determine trophic levels (Mearns 1982) carefully using well-known and accepted methods, such as stable isotope ratios, to assign quantitatively the position of an organism within a food web. Several acceptable techniques are available, including Cs:K ratio (Young and Mearns 1979; Schafer et al. 1982),  $^{13}\text{C}:^{12}\text{C}$  ( $^{13}\text{carbon}:^{12}\text{carbon}$ ) ratio (McConnaughey and McRoy 1979; Rau 1982), and  $^{15}\text{N}:^{14}\text{N}$  ( $^{15}\text{nitrogen}:^{14}\text{nitrogen}$ ) ratio (Rau 1982). The use of stable isotope ratios to assign position in a trophic web is based upon the observation that the relative proportion of the heavier isotope increases with trophic level. This may be due to preferential assimilation (i.e., uptake and retention) of the heavier isotope by a consumer, preferential loss of the lighter isotope from the consumer's tissues (e.g., through metabolism), or a combination of both (Rau 1982). As the diet of an organism may change dramatically during its life cycle, its position in the trophic web also may change. Aquatic organisms, thus, may appear to occupy many trophic levels simultaneously. Traditional methods of assigning level I to the producers (i.e., plants), II to herbivores, III to primary carnivores, etc., become confusing and untenable, particularly as the food habits of many species are not clearly defined. Assignment of trophic levels using isotopic ratio techniques seems to circumvent many of these problems, however.
- b. Sample all trophic levels including the microplankton, sediments, and water. The sampling should be uniform with respect to date and location and be done at regular time intervals to allow for seasonal effects. Sufficient numbers of organisms should be collected at each trophic level to ensure that the range of data values is representative of the species as a group. Samples also should be grouped by size class (within species), due to changes in position in the food web with age.
- c. Collect regular data on the physicochemical conditions (especially salinity, temperature, pH, alkalinity, turbidity, conductivity, organic matter, etc.) at each sampling station and date.
- d. Record information on size (weight and body length) and, if

possible, age (fish), sex, state of maturity, etc., for each species whenever possible.

- e. Obtain all data available from previous work at the same location. (The U.S. Geological Survey and other groups publish water resources data on nearly every major body of water in the United States, including lakes, streams, rivers, and estuaries.)
- f. Determine species of prey in stomachs of fish, if possible, and the concentrations of contaminants in the gut and feces. The weight of food and feces also should be determined whenever possible.

#### All studies

- a. Be sure the experiment is properly designed to answer the specific question. See a statistician before beginning, if necessary. Many of the papers reviewed herein did not answer the questions that were proposed, due to improper experimental design.
- b. Use an adequate number of specimens to ensure that data account for the natural variation within the population. When  $n = 1$ , no valid conclusions can be made about anything.
- c. Express the contaminant concentration in an organism on a whole-body, oven-dry-weight basis. Biomagnification, by definition, cannot be demonstrated using data reported on either an organ-by-organ or a tissue-fraction basis for one species, and whole-body basis for others. Exclude the gut contents, wherever possible, so as to include only the actual body burden. Using dry weight is very important, because of the inherent variability in water content both within and among species. Differences in water content may be the consequence of physiological and/or environmental changes, taxa-specific differences (i.e., those due to different proportions of hard tissues such as bone, carapace, shells, etc., in different taxonomic groups), and the loss of water during weighing and handling of specimens. The use of fresh weight may lead to gross errors when estimating contaminant concentrations in organisms. Based upon the data reviewed herein, trying to prove or disprove the biomagnification of lipid-soluble compounds (i.e., organic lipophilic compounds) based solely upon parts per million in lipids is unrealistic. Physiologically and toxicologically lipids or fats may be the most important reservoir of lipophilic compounds. Lipid (or fat) content, however, varies widely with season, species, age, body weight, and physiological condition of an organism. Also, the correlation between the body burden of contaminants and fat content is not always good. Consequently, the reporting of contaminant levels on a lipid basis sometimes may be misleading. If contaminant levels are to be specified on an organ, tissue, lipid fraction, wet weight, or some other basis, the oven-dry weights of these organs and the whole-body dry weights should be specified, in order to facilitate the use of the data on a comparative basis.



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Table 1  
Concentrations (ppm) of Heavy Metals in a Wastewater-Based Artificial Food Chain\*

Species	Cd	Cr	Cu	Ni	Pb	Sn	Zn
Algae							
<i>Scenedesmus</i> spp.	60 ± 12	13 ± 2.4	200 ± 40	15 ± 4	1,100 ± 285	70 ± 26	600 ± 40
Crustaceans							
<i>Daphnia magna</i>	5 ± 0.08	3 ± 0.3	60 ± 6	3 ± 0.6	40 ± 12	30 ± 3	300 ± 40
<i>Daphnia pulex</i>	2 ± 0.01	8 ± 1.2	30 ± 3	1.7 ± 0.19	6 ± 4.0	0.6 ± 0.2	120 ± 10
Fish							
<i>Notemigonus chrysoleucas</i>	--	--	2.54 ± 0.32	4.45 ± 0.36	4.91 ± 0.46	3.91 ± 0.36	64.36 ± 13.72
<i>Pimephales promelas</i>	--	--	3.27 ± 0.71	2.71 ± 0.40	4.05 ± 0.59	3.63 ± 0.34	80.76 ± 5.08
Water	0.002 ± 0.001	0.01 ± 0.002	0.02 ± 0.01	0.03 ± 0.01	0.03 ± 0.01	--	0.05 ± 0.005

\* Adapted from Tarifeño-Silva et al. (1982). Values are mean metal concentrations ± SE, expressed as parts per million dry weight.

Table 2  
Distribution of Cadmium and Lead in Jubilee Creek, Illinois\*

Species	Common Name	Feeding Habits	Cadmium, ppm		Lead, ppm	
			Mean $\pm$ 95% CI	Range	Mean $\pm$ 95% CI	Range
Aquatic Insects						
<i>Isonychia</i> sp.	Mayfly	Detritivore	1.19 $\pm$ 0.42	0.79 - 1.71	6.83 $\pm$ 2.78	3.97 - 11.76
<i>Agrion</i> sp.	Damselfly	Predator	1.54 $\pm$ 0.59	1.00 - 1.98	12.59 $\pm$ 2.03	11.22 - 15.09
<i>Cheumatopsyche</i> sp.	Caddisfly	Omnivore	0.81 $\pm$ 0.20	0.73 - 0.99	11.00 $\pm$ 3.43	7.98 - 13.02
<i>Hydropsyche</i> sp.	Caddisfly	Omnivore	0.53 $\pm$ 0.85	0.22 - 0.85	6.85 $\pm$ 2.49	3.53 - 11.57
Molluscs						
<i>Physa</i> sp.	Common pond snail	Detritivore	--	--	13.64 $\pm$ 4.97	9.50 - 18.96
Fishes						
<i>Carpoides carpio</i>	River carpsucker	Omnivore	0.08 $\pm$ 0.04	0.06 - 0.09	2.55 $\pm$ 2.31	1.98 - 3.62
<i>Etheostoma flabellare</i>	Fantail darter	Small predator	0.15 $\pm$ 0.04	0.08 - 0.32	2.88 $\pm$ 0.43	1.89 - 4.57
<i>Micropterus dolomieu</i>	Smallmouth bass	Predator	0.10 $\pm$ 0.07	0.05 - 0.19	2.47 $\pm$ 1.01	1.96 - 3.91
Sediments			0.17 $\pm$ 0.07	0.08 - 0.23	8.30 $\pm$ 1.97	6.18 - 11.45
Water			0.02		<0.5	

\* Adapted from Enk and Mathis (1977). Reprinted with permission from *Hydrobiologia*, Vol 52, p 156-157, Copyright 1977-- Dr. W. Junk Publishers.

Table 3  
Trace Metals in Aquatic Invertebrates from the Fox River, Illinois\*

Taxonomic Group	Genus	Mean Metal Concentration (ppm) $\pm$ 95% Confidence Limits			
		Cd	Cu	Pb	Zn
Insects					
Trichoptera (caddisflies)	<i>Hydropsyche</i>	1.52 $\pm$ 0.76	11.88 $\pm$ 0.56	18.81 $\pm$ 0.74	220.04 $\pm$ 0.38
	<i>Cheumatopsyche</i>	1.49 $\pm$ 0.46	14.10 $\pm$ 0.26	23.72 $\pm$ 0.63	270.82 $\pm$ 0.39
Ephemeroptera (mayflies)	<i>Stenonema</i>	5.55 $\pm$ 0.46	17.55 $\pm$ 0.54	30.04 $\pm$ 0.34	252.92 $\pm$ 0.11
	<i>Potamanthus</i>	6.33	12.96	39.48	229.62
	<i>Hexagenia</i>	$\leq 0.5$	11.29 $\pm$ 0.09	29.12 $\pm$ 0.26	177.83 $\pm$ 0.04
	<i>Baetis</i>	$\leq 0.5$	12.12	$\leq 4.0$	206.30
	<i>Simulium</i>	2.53	14.00	24.04	102.77
Diptera (flies)	Chironomidae	2.17	13.05	29.74	144.21
	<i>Argia</i>	$\leq 0.5$	74.21	23.71	183.41
Odonata (dragonflies and damselflies)	<i>Amphiagrion</i>	$\leq 0.5$	26.52	$\leq 4.0$	113.25
	<i>Anax</i>	$\leq 0.5$	27.79 $\pm$ 0.06	20.20 $\pm$ 0.06	75.52 $\pm$ 0.16
Hemiptera (bugs)**	<i>Sigara</i>	$\leq 0.5$	19.47 $\pm$ 0.13	19.51 $\pm$ 0.24	172.78 $\pm$ 0.08
	<i>Ranatra</i>	$\leq 0.5$	15.75	$\leq 4.0$	103.40
	<i>Belostoma</i>	$\leq 0.5$	18.50	$\leq 4.0$	228.01
	<i>Notonecta</i>	$\leq 0.5$	20.09	$\leq 4.0$	163.52
	<i>Helophorus**</i>	$\leq 0.5$	16.22	$\leq 4.0$	119.26
	<i>Berosus</i>	$\leq 0.5$	$\leq 1.0$	$\leq 4.0$	160.92
	<i>Tropisternus</i>	$\leq 0.5$	$\leq 1.0$	$\leq 4.0$	170.51
Coleoptera (beetles)	<i>Dineutus</i>	$\leq 0.5$	$\leq 1.0$	$\leq 4.0$	107.50
Crustaceans					
Decapoda	<i>Orconectes</i>	1.60 $\pm$ 0.50	86.61 $\pm$ 0.45	25.68 $\pm$ 0.53	107.12 $\pm$ 0.24
	<i>Procambarus</i>	2.77 $\pm$ 0.04	58.10 $\pm$ 0.05	15.73 $\pm$ 0.01	64.68 $\pm$ 0.04
	<i>Cambarus</i>	1.74 $\pm$ 0.05	94.76 $\pm$ 0.04	15.62 $\pm$ 0.01	93.43 $\pm$ 0.03
(Continued)					

(Continued)

\* Adapted from Anderson (1977). Reprinted with permission from the Bulletin of Environmental Contamination and Toxicology, Vol 18, pp 347-348, Copyright 1977--Springer-Verlag. Values expressed as parts per million dry weight.  
 \*\* Indicates adult stage of life cycle.

Table 3 (Concluded)

Taxonomic Group	Genus	Mean Metal Concentration (ppm) $\pm$ 95% Confidence Limits			
		Cd	Cu	Pb	Zn
Crustaceans (Continued)					
Isopoda	<i>Asellus</i>	2.62 $\pm$ 0.66	99.19 $\pm$ 0.11	22.05 $\pm$ 0.38	124.94 $\pm$ 0.39
Amphipoda	<i>Gammarus</i>	<0.05	70.74	<4.0	101.19
Molluscs					
Gastropoda (snails)	<i>Physa</i>	2.97 $\pm$ 0.44	22.01 $\pm$ 0.21	21.04 $\pm$ 0.31	69.93 $\pm$ 0.74
	<i>Campeloma</i>	1.76	18.37	21.79	99.58
	<i>Goniobasis</i>	2.19 $\pm$ 0.37	13.40 $\pm$ 0.29	19.73 $\pm$ 0.48	22.69 $\pm$ 0.24
	<i>Pleurocerca</i>	2.31 $\pm$ 0.03	10.70 $\pm$ 0.12	24.08 $\pm$ 0.03	19.19 $\pm$ 0.02
Pelecypoda (clams)	<i>Sphaerium</i>	1.99 $\pm$ 0.19	10.06 $\pm$ 0.41	32.18 $\pm$ 0.28	61.07 $\pm$ 0.19
	<i>Lampsilis</i>	2.23 $\pm$ 0.27	12.67 $\pm$ 0.13	21.93 $\pm$ 0.14	353.04 $\pm$ 0.10
	<i>Anodonta</i>	1.78	6.88	13.73	232.10
	<i>Lasmigona</i>	1.43	5.15	23.70	317.63
	<i>Strophitus</i>	2.52	8.68	27.25	208.80
	<i>Anodonta*</i>	1.35 $\pm$ 0.26	9.30 $\pm$ 0.14	10.19 $\pm$ 0.11	3.70 $\pm$ 0.11
Annelids					
Hirudinea (leeches)	<i>Erpobdella</i>	3.80 $\pm$ 0.30	16.83 $\pm$ 0.36	39.78 $\pm$ 0.08	136.23 $\pm$ 0.54
	<i>Placobdella</i>	<0.5	7.59	<4.0	148.37

\* Indicates shell only.

Table 4  
Heavy Metals in Water (ppb) and Biota (ppm) from Coastal Areas of the British Isles in 1970\*

Organism	Ag	Cd	Cu	Ni	Pb	Zn
<b>Area 2</b>						
<b>Algae</b>						
<i>Fucus</i> sp.	0.24 (0.13 - 0.47)	1.1 (0.5 - 1.5)	4.6 (2.4 - 9.3)	4.3 (2.9 - 9.5)	3.2 (1.4 - 7.7)	94 (42 - 450)
<i>Porphyra</i> sp.	0.13 (0.01 - 0.30)	0.35 (0.10 - 0.97)	8.9 (2.8 - 23.3)	2.2 (0.2 - 9.6)	3.1 (0.8 - 10.5)	63 (36 - 174)
<b>Limpet</b>						
<i>Patella</i>	1.3 (0.6 - 2.7)	8.4 (2.8 - 35)	9.9 (5.5 - 20.0)	7.3 (3.1 - 24)	7.8 (3.5 - 85)	90 (56 - 195)
Water (filtered)	0.08 (0.02 - 0.24)	0.41 (0.03 - 1.43)	1.4 (0.9 - 2.7)	1.4 (0.9 - 3.1)	1.6 (0.9 - 2.9)	6.6 (4.9 - 11.1)
<b>Area 3</b>						
<b>Algae</b>						
<i>Fucus</i> sp.	0.30 (0.07 - 0.79)	1.4 (0.5 - 3.0)	9.0 (3.7 - 16.9)	6.7 (5.0 - 9.8)	3.4 (0.6 - 9.0)	171 (88 - 962)
<i>Porphyra</i> sp.	0.09 (0.01 - 0.21)	0.25 (0.05 - 0.87)	11.5 (6.6 - 19.5)	2.0 (0.6 - 9.7)	2.7 (1.1 - 10.5)	66 (35 - 177)
<b>Limpets</b>						
<i>Patella</i> sp.	2.1 (1.3 - 3.6)	13.1 (3.8 - 23)	14.4 (9.5 - 22.0)	7.0 (4.5 - 9.9)	7.9 (5.4 - 12.5)	158 (109 - 274)
Water (filtered)	0.04 (0.03 - 0.16)	0.46 (0.15 - 1.14)	1.7 (1.1 - 3.1)	2.6 (1.3 - 9.8)	1.3 (0.5 - 2.4)	6.8 (3.8 - 49.1)

\* Adapted from Preston et al. (1972). Values are expressed as parts per million dry weight for organisms and parts per billion (micrograms per liter) in water.

Table 5

Trace Metals (ppm) in Pelagic Organisms from the Northeast Atlantic Ocean\*

Species	Common Name	Feeding Habits	As	Cd	Hg	Zn
<b>Coelenterates</b>						
<i>Pelagia</i> sp.	Jellyfish	Predator	11	5.3	0.07	28
<b>Crustaceans</b>						
<i>Labidocera acutifrons</i>	Copepod	Filter-feeder	14.5	9.8	0.12	--
<i>Eucopi sculpticauda</i>	Mysid	Filter-feeder	30	2.0	0.20	--
<i>Meganytiophanes norvegica</i>	Euphausiid	Omnivore	42	0.25	0.26	104
<i>Systellapsis debilis</i>	Decapod	--	22	13	0.22	50
<i>Oplophorus</i> sp.	Decapod	--	23.5	13	0.36	98
<i>Acanthophyra oximia</i>	Decapod	--	17	3.0	0.36	65
<b>Tunicate</b>						
<i>Pyrosoma</i> sp.	Tunicate	Herbivore	1.5	0.44	0.06	105
<b>Fish</b>						
<i>Hygrcphum macrochir</i>			2.5	0.98	0.25	--
<i>Diaphus dumerili</i>	Headlight fish	Small predator	2.7	0.73	0.12	44
<i>Scombresox saurus</i> (muscle)	Skipper	Predator?	5.4	0.05	0.20	--
(heart)			6.6	0.10	0.21	--
(liver)			8.4	0.62	0.22	--
<i>Etmopterus spinax</i> (liver)	Shark	Top predator	1.2 - 1.3	--	0.04 - 0.18	--

\* Adapted from Leatherland et al. (1973). Values are expressed as parts per million dry weight.

Table 6  
Trace Element Concentrations (ppm) in Marine Organisms of  
Card Sound, Florida\*

Species	Cu	Zn	Cd	Pb
<b>Macrophyta</b>				
<i>Thalassia testudinum</i>	1.6 ± 0.33	18 ± 1.3	0.20 ± 0.021	0.72 ± 0.16
<i>Laurencia poitei</i>	12 ± 2.4	34 ± 5.1	0.20 ± 0.047	0.59 ± 0.16
<i>Penicillus capitatus</i>	1.2 ± 0.17	12 ± 3.5	0.11 ± 0.012	1.1 ± 0.21
<i>Halimeda incrassata</i>	0.70 ± 0.26	3.7 ± 1.2	0.16 ± 0.12	1.2 ± 0.56
<i>Rhizophora mangle</i>				
Leaves (live)	1.3 ± 0.67	3.1 ± 0.88	0.044 ± .028	0.39 ± 0.11
(dead)	5.8 ± 4.6	2.3 ± 0.52	0.24 ± 0.11	0.79 ± 0.23
Seedlings in water	0.81 ± 0.79	2.2 ± 0.58	0.017 ± .0059	0.23 ± 0.17
Decaying stems in water	0.52 ± 0.46	8.1 ± 5.9	0.056 ± 0.055	0.099 ± 0.0072
<b>Microphyta</b>				
Phytoplankton	12 ± 8.0	180 ± 80	0.20	0.33
Epiphytes on <i>Thalassia</i> blades				
<b>Macrofauna</b>				
Detritivores and Carnivores	7.4 ± 0.67	28 ± 20	0.19 ± 0.08	0.39 ± 0.15
Sponges	3.7 ± 1.5	24 ± 9.8	0.44 ± 0.18	0.36 ± 0.15

\* Adapted from Gilio and Segar (1976). Reprinted with permission of the authors. Values are expressed as mean metal concentration ± standard error of the means on the basis of ppm dry weight.



Table 7  
Heavy Metal Concentrations (ppm) in Biota from Cockburn Sound, Australia\*

Species	Common Name	Cd		Cu		Pb		Zn	
		$\bar{X}$	Range	$\bar{X}$	Range	$\bar{X}$	Range	$\bar{X}$	Range
Site 11									
<i>Chaetopterus rariopedatus</i>	Mussel	--	0.3	--	2.4 - 2.5	--	1.0	--	25 - 43
<i>Portunus pelagicus</i>	Polychaete Crab	0.4	--	2.2	--	--	--	20.3	--
	Hepatopancreas	8.0	3.8 - 13.8	19.9	5.2 - 42.7	1.5	0.8 - 2.1	50.3	30.7 - 64.9
	Flesh	0.3	0.3 - 0.4	3.9	1.5 - 5.3	0.9	0.7 - 1.3	21.5	17.0 - 32.2
<i>Ulva lactuca</i> **	Sea Lettuce	0.3	--	4.6	--	0.4	--	30.4	--
Site 23									
<i>Chaetopterus rariopedatus</i>	Mussel	--	5.5 - 5.8	--	13.7 - 13.8	--	2.2 - 2.8	--	135 - 149
<i>Portunus pelagicus</i>	Polychaete Crab	4.8	--	29.7	--	--	--	34.8	--
	Hepatopancreas	26.6	10.8 - 68.2	15.2	7.3 - 33.7	3.4	2.0 - 5.7	100.9	29.2 - 239
	Flesh	0.3	0.1 - 0.6	5.6	3.5 - 10.1	1.2	0.6 - 2.0	18.8	8.6 - 30.7
Site 28 (N)									
<i>Ostrea angasi</i>	Mussel	--	1.8 - 2.0	--	1.9 - 2.0	--	0.8 - 1.8	--	17 - 25
<i>Chaetopterus rariopedatus</i>	Oyster	--	1.5 - 8.7	--	3.6 - 40.1	--	0.8 - 3.2	--	126 - 1190
<i>Portunus pelagicus</i>	Polychaete Crab	1.5	--	1.9	--	--	--	17	--
	Hepatopancreas	18.5	3.0 - 52.2	22.0	7.3 - 32.9	4.5	0.5 - 7.5	101.9	41.6 - 256
	Flesh	0.3	0.2 - 0.5	6.5	2.0 - 9.5	1.4	0.2 - 2.1	20.7	15.0 - 24.5
<i>Posidonia</i> sp.	Seagrasses	0.4	--	3.5	--	--	--	67.2	--
Epiphytes on <i>Posidonia</i>	--	0.5	--	6.3	--	--	--	38.0	--
Site 29									
<i>Chaetopterus rariopedatus</i>	Mussel	--	0.3 - 0.6	--	1.7 - 1.8	--	1.1 - 1.7	--	31 - 34
<i>Portunus pelagicus</i>	Polychaete Crab	0.6	--	1.1	--	--	--	7.7	--
	Hepatopancreas	15.2	4.1 - 37.9	9.7	4.1 - 33	4.1	1.6 - 10	80.2	31.9 - 151
	Flesh	0.7	0.4 - 1.5	3.2	1.3 - 6.1	0.9	0.4 - 1.4	19.1	14.0 - 24.2
<i>Posidonia</i> sp.††	Seagrasses	0.2	--	1.8	--	--	--	56.2	--
Epiphytes on <i>Posidonia</i> ††	--	0.4	--	7.6	--	--	--	36.4	--

\* Adapted from Talbot and Chegwiddden (1982).

\*\* Values reported as micrograms per gram dry weight. Data were taken from site "W", adjacent to site 11.

† The ranges for oyster are the overall minimum and maximum values for several collecting dates at the same location.

†† Data were taken from site "H", adjacent to site 29.

Table 8  
Heavy Metal Concentrations (ppm) in Organisms from the Loire Estuary (France) in Relation to Feeding Habits\*

Species	Portion Analyzed	Feeding Habits**	Predominant Food**	Cd	Pb	Cu	Zn
Banc de Bilho							
Ctenophores							
<i>Nemopsis bachei</i>	whole	--	--	0.496 ± 0.006	0.368 ± 0.128	0.687 ± 0.367	47.9 ± 3.9
Crustaceans							
<i>Neomysis integer</i>	whole	--	--	0.069 ± 0.006	4.652 ± 0.239	20.492 ± 0.872	47.8 ± 1.0
<i>Mesopodopsis slabberi</i>	whole	--	--	0.142 ± 0.011	5.748 ± 0.271	7.370 ± 0.288	62.8 ± 7.9
<i>Crangon crangon</i>	whole	Omnivorous	<i>N. integer</i>	0.103 ± 0.003	2.520 ± 0.101	22.425 ± 1.413	39.0 ± 0.2
<i>Carcinus maenas</i>	whole	--	--	0.289 ± 0.036	0.662 ± 0.051	30.835 ± 5.299	68.5 ± 6.4
Fishes							
<i>Solea solea</i>	whole†	Sediments	<i>N. integer</i>	0.047 ± 0.003	0.711 ± 0.056	3.739 ± 0.342	57.8 ± 5.0
	digestive contents			0.412 ± 0.009	31.935 ± 1.518	5.676 ± 0.274	125.5 ± 1.9
<i>Platichthys flesus</i>	whole†	Sediments	<i>N. integer</i>	0.039 ± 0.010	1.411 ± 0.133	--	114.5 ± 10.0
	digestive contents			0.421 ± 0.015	51.082 ± 2.379	18.044 ± 0.455	123.6 ± 1.3
<i>Dicentrarchus labrax</i>	whole†	Omnivorous	<i>N. integer</i>	0.116 ± 0.004	0.584 ± 0.006	3.306 ± 0.069	81.5 ± 1.3
	stomach contents			0.568 ± 0.007	5.894 ± 0.146	24.012 ± 1.179	83.5 ± 1.4
	intestine contents			1.047 ± 0.059	12.324 ± 0.560	35.283 ± 1.871	211.6 ± 5.7
<i>Gobius microps</i>	whole†	Omnivorous	<i>N. integer</i>	0.044 ± 0.008	0.360 ± 0.100	1.203 ± 0.147	58.0 ± 3.9
	digestive contents			0.255 ± 0.017	19.012 ± 2.578	3.965 ± 0.770	111.3 ± 2.3
<i>Osmerus eperlanus</i>	whole†	Planktivorous	<i>N. integer</i>	0.009 ± 0.005	1.929 ± 0.494	2.243 ± 0.568	53.5 ± 3.4
(small)	digestive contents			0.013 ± 0.004	4.944 ± 0.403	3.188 ± 0.376	58.7 ± 0.4
<i>Osmerus eperlanus</i>	whole†	Planktivorous	<i>N. integer</i>	0.022 ± 0.001	1.385 ± 0.058	0.892 ± 0.150	81.5 ± 1.3
(large)	digestive contents			0.046 ± 0.001	6.135 ± 0.059	13.941 ± 1.422	69.7 ± 2.4
<i>Sprattus sprattus</i>	whole†	Planktivorous	<i>N. slabberi</i>	0.139 ± 0.006	0.238 ± 0.023	3.542 ± 0.983	119.6 ± 4.3
(small)							
<i>Sprattus sprattus</i>	whole†			0.025 ± 0.002	1.133 ± 0.173	1.444 ± 0.232	68.5 ± 2.3
(large)	stomach contents			0.036 ± 0.003	2.483 ± 0.142	6.262 ± 0.263	49.2 ± 4.2
	intestine contents			0.227 ± 0.008	4.795 ± 0.359	13.496 ± 1.565	146.6 ± 5.7

(Continued)

\* Adapted from Amiard et al. (1980). Reprinted with permission from Water Research, Vol 14, pp 669-670, J. C. Amiard, C. Amiard-Triquet, C. Metayer, J. Marchand, and R. Ferre, Etude du Transfert de Cd, Pb, Cu, et N dans les Chaînes Trophiques Maritimes et Estuariennes - I. Etat dans L'Estuaire Interne de la Loire (France) au Cours de L'Ete 1978, Copyright 1980--Pergamon Press, Ltd. Values are mean metal concentrations expressed as parts per million dry weight ± standard deviation.

\*\* Feeding habits and predominant foods as designated by authors.

† Whole fish, digestive contents excluded.

Table 8 (Concluded)

Species	Portion Analyzed	Feeding Habits	Predominant Food	Cd	Pb	Cu	Zn
<i>Engraulis encrasicolus</i>	whole*	Planktivorous	<i>N. integer</i> and	0.068 ± 0.008	0.746 ± 0.158	3.237 ± 0.430	81.3 ± 5.4
	stomach contents		<i>N. slabberi</i>	0.156 ± 0.007	6.544 ± 0.179	19.744 ± 1.298	93.6 ± 0.6
	intestine contents			0.549 ± 0.014	18.183 ± 1.548	35.000 ± 1.121	283.9 ± 1.0
<i>Syngnathus rostellatus</i>	whole*	Planktivorous	<i>N. integer</i>	0.123 ± 0.017	1.616 ± 0.071	1.576 ± 0.316	90.0 ± 3.0
	eggs			<0.010	0.837 ± 0.102	3.609 ± 0.160	98.3 ± 0.5
	digestive contents			0.109 ± 0.015	6.621 ± 0.272	20.073 ± 1.508	67.1 ± 1.6
<i>Merlangius merlangus</i>	whole*	Top carnivore	<i>N. integer</i>	0.037 ± 0.003	0.693 ± 0.045	1.261 ± 0.137	50.1 ± 2.7
	stomach contents			0.081 ± 0.013	4.463 ± 0.536	19.139 ± 0.883	60.1 ± 2.4
Banc de Pipy							
Ctenophores							
<i>Nemopsis bachei</i>	whole	--	--	0.602 ± 0.031	0.635 ± 0.067	2.999 ± 0.146	69.9 ± 2.8
Crustaceans							
<i>Crangon crangon</i>	whole	Omnivorous	Oligochaetes	0.327 ± 0.008	2.715 ± 0.184	68.830 ± 6.535	74.7 ± 2.1
<i>Palaemon longirostris</i>	whole	Omnivorous	Polychaetes	0.330 ± 0.010	2.337 ± 0.214	145.203 ± 3.652	81.6 ± 1.1
Fishes							
<i>Platichthys flesus</i>	whole*	Sediments	<i>Boccardia ligierica</i>	0.040 ± 0.003	1.237 ± 0.078	2.944 ± 0.170	131.4 ± 7.0
	digestive contents			1.489 ± 0.020	34.205 ± 1.465	21.575 ± 3.848	166.9 ± 3.2
<i>Dicentrarchus labrax</i>	whole*	Omnivorous	Polychaetes	0.084 ± 0.011	0.381 ± 0.069	4.298 ± 0.434	87.7 ± 5.3
	digestive contents			1.070 ± 0.063	36.777 ± 2.291	20.755 ± 1.548	144.5 ± 1.0
<i>Gobius microps</i>	whole*	Omnivorous	<i>N. integer</i> and	<0.010	0.903 ± 0.073	1.159 ± 0.062	55.8 ± 3.0
	digestive contents		<i>N. slabberi</i>	0.226 ± 0.018	20.733 ± 2.656	29.462 ± 5.270	135.0 ± 10.6
<i>Osmerus eperlanus</i> (small)	whole*	Planktivorous	<i>N. integer</i>	0.017 ± 0.007	3.072 ± 0.170	1.450 ± 0.137	84.3 ± 7.9
	digestive contents			0.030 ± 0.005	8.701 ± 0.156	13.778 ± 0.438	77.7 ± 0.3
<i>Osmerus eperlanus</i> (large)	whole*			<0.010	2.659 ± 0.360	1.242 ± 0.129	62.7 ± 6.1
	digestive contents			0.066 ± 0.009	4.447 ± 0.147	12.167 ± 2.844	65.6 ± 3.1

\* Whole fish, digestive contents excluded.

Table 9  
Concentrations (ppm) of Heavy Metals in Sediments and Biota in the Looe Estuary (Cornwall), U. K., in 1975-1976\*

Species	Common Name	Feeding Habits	Ag	Cd	Cr	Cu	Ni	Pb	Zn
<b>Algae</b>									
<i>Fucus</i> sp.	brown algae	--	0.82 (0.24 - 2.42)	1.30 (0.86 - 2.41)	2.24 (0.56 - 3.54)	17 (3.5 - 33)	9.7 (5.7 - 13.6)	38 (3.1 - 86)	198 (56 - 340)
<b>Molluscs</b>									
<i>Littorina littorea</i>	periwinkle	herbivore	19.6 (3.2 - 73)	1.38 (0.49 - 2.56)	0.64 (0.13 - 0.98)	124 (62 - 194)	3.1 (2.2 - 4.1)	19 (3.7 - 70)	117 (45 - 284)
<i>Patella vulgata</i>	limpet	herbivore	3.0 (1.5 - 6.0)	8.6 (3.3 - 21.5)	1.38 (0.48 - 2.62)	19 (10 - 27)	2.5 (1.7 - 3.7)	25 (5.1 - 38)	165 (83 - 224)
<i>Cerastoderma edule</i>	cockle	filter feeder	1.52 (0.11 - 6.5)	0.84 (0.18 - 1.04)	2.04 (1.34 - 2.46)	9.8 (5.2 - 27.2)	44 (34 - 62)	8.1 (4.7 - 15.6)	55 (46 - 66)
<i>Mytilus edulis</i>	mussel	filter feeder	0.23 (0.10 - 0.55)	1.78 (0.84 - 2.64)	1.92 (0.94 - 2.74)	9.5 (3.9 - 13.6)	2.2 (0.9 - 3.5)	54 (30 - 105)	132 (57 - 199)
<i>Scrobicularia plana</i>	clam	deposit feeder	40 (1.1 - 185)	1.62 (0.60 - 3.37)	2.77 (1.08 - 3.89)	133 (16 - 365)	9.4 (5.3 - 13.9)	189 (62 - 473)	974 (606 - 1600)
<i>Macoma balthica</i>	clam	deposit feeder	85 (19 - 128)	0.67 (0.21 - 0.85)	2.52 (1.89 - 3.30)	300 (96 - 615)	7.5 (6.9 - 7.9)	34 (15 - 61)	804 (510 - 1160)
<i>Mucella lapillus</i>	dogwhelk	predator	2.7 (1.3 - 4.2)	12.8 (5.5 - 16.0)	2.16 (0.39 - 5.61)	110 (51 - 141)	2.3 (1.4 - 4.1)	5.1 (1.9 - 7.1)	416 (235 - 520)
<b>Annelid</b>									
<i>Nereis diversicolor</i>	polychaete worm	deposit feeder	5.2 (0.7 - 30)	0.53 (0.1 - 3.4)	0.55 (0.1 - 2.39)	44 (22 - 78)	3.3 (2.1 - 5.2)	45 (2.1 - 261)	215 (170 - 258)
<b>Sediments</b>									
			1.5 (0.17 - 7.3)	0.2 (0.1 - 0.54)	36 (22 - 55)	63 (13 - 178)	34 (23 - 51)	280 (18 - 2030)	151 (47 - 250)

\* Adapted from Bryan and Hummerstone (1977). Reprinted with permission from the Journal of the Marine Biological Association of the United Kingdom, Vol 57, p 83, 88, Copyright 1977--Cambridge University Press. Values are mean metal concentrations expressed as parts per million dry weight, with ranges shown in parentheses.

Table 10  
Concentrations (ppm) of Heavy Metals in Two Marine and One Inland Saltwater Food Webs\*

Species	Common Name	Estimated Trophic Level	Cs:K Ratio x 10 <sup>-6</sup>	Ag	Cd	Cr	Cu	Organic Total Hg	Ni	Pb	Zn
<u>Salton Sea Food Web (Inland)</u>											
<i>Cynoscion xanthulus</i>	Orangemouth corvina	IV-V	32.0	<0.003	<0.003	<0.016	0.30	0.030	0.016	<0.04	3.1
<i>Bairdiella icistia</i>	Gulf croaker	III-IV	20.9	0.002	0.001	0.018	0.46	0.009	0.009	<0.04	3.2
<i>Anisotremus elavidsoni</i>	Sargo	III-IV	18.8	0.002	0.001	0.028	0.62	0.012	0.005	<0.03	3.5
<i>Dorosoma petenense</i>	Threadfin shad	III	10.0	<0.002	<0.001	<0.010	1.3	--	--	<0.04	3.9
<i>Poecilia latipinna</i>	Sailfin molly	II-III	14.3	0.002	0.002	0.030	0.30	0.008	0.005	<0.02	5.3
<u>Newport Bay Food Web</u>											
<i>Morone saxatilis</i>	Striped bass	IV-V	4.94	0.003	0.003	<0.009	0.27	0.36	0.41	<0.03	4.1
<i>Paralabrax maculatofasciatus</i>	Spotted sand bass	IV-V	5.51	0.003	0.003	0.014	0.26	0.27	0.20	<0.04	4.3
<i>Umbrina roncadore</i>	Yellowfin croaker	III-IV	5.53	0.003	0.002	0.008	0.26	0.054	0.050	<0.03	5.8
<i>Atherinops affinis</i>	Topsmelt	III	3.69	0.002	0.002	<0.010	0.20	0.092	0.051	<0.04	14
<i>Mulgil cephalus</i>	Striped mullet (large) (small)	II II	4.47 3.59	0.002 0.002	0.020 <0.002	0.016 0.018	0.55 0.24	0.017 0.014	0.010 0.017	<0.04 <0.04	3.3 2.9
<u>Palos Verdes Food Web</u>											
<u>Fishes</u>											
<i>Sebastes paucispinis</i>	Bocaccio	IV-V	16.6	0.008	<0.002	<0.010	0.15	--	0.14	0.058	4.7
<i>Scorpaena guttata</i>	Scorpionfish	IV-V	13.6	0.022	0.004	0.036	0.15	--	0.38	0.15	3.9
<i>Citharichthys sordidus</i>	Sanddab	III-IV	12.1	0.005	0.003	0.032	0.19	--	0.081	0.056	3.2
<u>Crustaceans</u>											
<i>Cancer anthonyi</i>	Yellow crab	III-IV	6.5	0.095	0.004	0.080	7.84	--	0.064	0.14	25.2
<i>Sicyonia ingentis</i>	Ridgeback prawn	III-IV	11.2	<0.004	0.032	<0.019	2.0	--	0.080	<0.01	9.8
<u>Molluscs</u>											
<i>Hinnites multirugosus</i>	Scallop	II-III	5.4	<0.003	0.803	0.255	0.24	--	0.056	0.046	19.8
<i>Haliotis cracherodii</i>	Black abalone	II	7.6	0.028	0.041	0.95	3.35	--	0.010	0.68	6.1

\* Adapted from Young and Mearns (1979). Values are mean metal concentrations expressed as parts per million wet weight of muscle tissues.

Table 11  
Concentrations (ppm) of Heavy Metals in Animals in Three Marine Food Webs\*

Species	Trophic Level	Cs:K Ratio x 10 <sup>6</sup>	Coastal Pelagic Food Web										
			Ag	As	Cd	Cr	Cu	Organic Hg	Total Hg	Ni	Pb	Se	Zn
White shark	5.02	31.7	<0.002	3.18	0.004	0.145	0.7	8.15	8.10	<0.019	<0.056	0.45	5.05
Mako shark	4.40	19.7	0.006	3.53	0.043	0.075	0.27	1.47	1.44	<0.037	<0.12	0.53	5.3
Sea lion	4.02	10.7	--	0.25	--	--	--	0.367	0.316	--	--	0.57	--
Blue shark	4.00	13.7	--	--	--	--	--	0.506	0.710	--	--	--	--
Swordfish	3.97	12.3	<0.002	--	0.078	0.018	0.35	1.20	2.18	<0.021	<0.064	--	6.44
Thresher shark	3.82	25.5	<0.003	--	0.010	0.072	0.226	0.642	0.661	<0.072	<0.124	--	4.05
Bonita	3.80	8.79	<0.003	0.27	0.010	0.037	0.481	0.210	0.207	<0.071	<0.12	2.11	3.91
Barracuda	3.74	4.20	--	--	--	--	--	0.197	0.274	--	--	--	--
Pacific mackerel	3.54	6.86	0.003	0.54	0.059	0.032	0.37	0.086	0.118	0.091	<0.10	0.58	4.4
Market squid	3.62	3.32	0.120	0.68	0.496	0.037	15.8	0.041	0.026	<0.018	<0.053	2.16	9.72
Pacific hake	3.09	8.58	<0.002	--	0.003	0.073	0.20	0.061	0.108	<0.021	<0.063	--	2.59
Jack mackerel	3.04	5.73	--	0.46	--	--	--	0.049	0.046	--	--	1.15	--
Sardine	3.01	4.02	<0.003	0.81	0.021	0.035	0.335	0.014	0.048	<0.082	<0.153	0.78	3.94
Basking shark	3.00	16.1	--	--	--	--	--	0.065	0.096	--	--	--	--
Blue whale	3.00	11.0	--	0.19	--	--	--	0.034	0.021	--	--	0.68	--
Anchovy	2.82	1.87	0.010	1.66	0.172	0.068	0.39	0.022	0.033	<0.038	<0.12	0.54	7.6
Zooplankton	2.0	3.2	0.044	1.32	0.83	0.114	1.12	<0.004	<0.034	0.294	0.384	0.48	10.5
Palos Verdes Epibenthic Food Web													
Scorpionfish	4.53	6.22	<0.002	--	<0.006	<0.015	0.194	0.247	0.247	<0.030	<0.062	--	2.54
Spiny dogfish	4.16	31.3	<0.002	--	0.018	0.031	0.097	1.48	1.53	<0.021	0.058	--	3.85
Dover sole	3.52	3.62	<0.002	--	0.006	<0.018	0.181	0.024	0.046	<0.020	<0.059	--	2.86
White croaker	3.36	2.85	<0.002	--	<0.002	<0.012	0.240	0.063	0.071	<0.023	<0.064	--	2.10
Ridgeback prawn	3.33	3.04	<0.001	--	0.12	0.188	5.49	0.047	0.072	<0.019	<0.056	--	12.0
Myxids and decapods	2.78	13.8	0.167	--	0.108	1.17	5.41	--	0.006	1.07	0.330	--	9.11
Sediments	--	840	11.4	--	27.1	802	387	--	--	73	223	--	993

(Continued)

\* Adapted from Schafer et al. (1982). Values are mean metal concentrations expressed as parts per million wet weight.

Table 11 (Concluded)

Species	Trophic Level	Cs:K Ratio $\times 10^6$	Ag	As	Cd	Cr	Cu	Organic Hg	Total Hg	Ni	Pb	Se	Zn
Eastern Tropical Pacific Food Web (Open Ocean)													
Yellowfin tuna (50 kg)	4.82	13.3	--	--	--	--	--	--	0.88	--	--	--	--
Silky shark	4.81	22.8	--	4.10	--	--	--	1.47	1.22	--	--	0.96	--
Skipjack tuna	4.44	8.59	--	0.64	--	--	--	0.041	0.229	--	--	0.92	--
Yellowfin tuna (4 kg)	4.29	12.7	--	0.49	--	--	--	0.284	0.157	--	--	0.85	--
Frigate tuna	3.92	8.39	--	--	--	--	--	0.402	0.296	--	--	--	--
Squid	3.52	1.94	--	2.60	--	--	--	0.018	0.051	--	--	0.61	--
Flying fish	3.00	7.00	--	0.38	--	--	--	0.045	0.010	--	--	0.92	--
Zooplankton	2.00	3.30	--	0.77	--	--	--	<0.003	0.016	--	--	0.82	--

Table 12  
Concentrations (ppm) of Heavy Metals in Leaf Detritus, Tipula sp.  
Larvae, and Feces, and Percent Body Burdens Associated  
with Gut Contents\*

<u>Metal</u>	<u>Leaf Detritus</u>	<u>Tipula larvae</u>		<u>Feces</u>	<u>% Body Burden Associated With Gut Contents</u>
		<u>Before Gut Evacuation</u>	<u>After Gut Evacuation</u>		
Cr	36.0 ± 3.8	13.1 ± 1.0	14.2 ± 0.5	39.39 ± 0.67	40.1 ± 2.5
Hg	0.13 ± 0.03	0.17 ± 0.05	0.22 ± 0.06	2.155 ± 0.084	73.8 ± 10.0
Se	0.49 ± 0.15	1.02 ± 0.13	0.95 ± 0.11	0.987 ± 0.019	22.9 ± 3.0
Zn	36.5 ± 9.6	90.1 ± 6.6	106.4 ± 4.7	763.9 ± 216.5	66.4 ± 10.8

\* Adapted from Elwood, Hildebrand, and Beauchamp (1976). Values are mean metal concentrations expressed as parts per million dry weight ± 2 standard errors of the means.



Table 13  
Mean Concentrations (ppm) of Toxic Trace Metals in Water, Sediments, and  
Biota of a Coal Ash Basin in South Carolina Before and After Dredging\*

Metal	Component				
	Water	Sediments	Plants	Invertebrates	Vertebrates
As	0.06 (0.07)	20 (27)	4.2 (5.3)	2.1 (60)	0.5 (4.0)
Cd	0.12 (0.09)	1.7 (3.8)	1.5 (0.9)	4.0 (2.5)	1.3 (0.9)
Cr	0.16 (0.20)	38 (34)	5.7 (2.9)	9.7 (3.5)	2.8 (5.8)
Cu	0.39 (0.40)	31 (32)	7.2 (14)	31 (67)	12 (38)
Hg	0.03 (0.02)	0.8 (0.8)	0.4 (0.4)	0.5 (0.2)	0.2 (0.3)
Se	0.10 (0.20)	6.1 (11)	1.8 (5.0)	2.6 (6.5)	9.4 (8.4)
Zn	0.39 (3.30)	6.4 (7.6)	5.0 (51)	15 (25)	12 (67)

\* Adapted from Cherry and Guthrie (1977). Reprinted with permission from Water Resources Bulletin, Vol 13, p 1230, Copyright 1977--American Water Resources Association. Values shown are mean metal concentrations expressed as parts per million (authors did not designate whether on a wet or dry weight basis). First value shown is before dredging; value in parentheses is after dredging.

Table 14  
Heavy Metals in Sediments, Water, and Biota of the Illinois River\*

Species	Feeding Habit	Cd	Cr	Cu	Ni	Pb	Zn
<b>Molluscs (clams)</b>							
<i>Fusconaia flava</i>	Filter feeders	0.69 (0.36-1.17)	7.7 (1.1-11.6)	1.7 (0.9-2.0)	2.1 (0.7-3.0)	3.7 (1.8-5.1)	66 (25-120)
<i>Anodonta plicata</i>	Filter feeders	0.38 (0.15-1.41)	4.4 (0.6-9.9)	1.2 (0.3-3.2)	1.1 (0.4-2.3)	2.7 (1.1-7.6)	95 (40-178)
<i>Quadrula quadrula</i>	Filter feeders	0.56 (0.31-1.37)	4.7 (1.8-8.3)	1.7 (1.1-3.6)	0.9 (0.4-1.6)	2.2 (0.5-3.8)	48 (28-64)
<i>Annelids (Tubificids)</i>	Detritivore	1.1 (0.5-3.2)	10 (4-21)	23 (10-42)	11 (4-18)	17 (6-39)	41 (13-62)
<b>Fishes</b>							
<i>Ichthyobus cyprinellus</i>	Omnivorous	0.032 (0.001-0.055)	0.13 (0.02-0.53)	0.18 (0.07-0.26)	0.10 (0.02-0.18)	0.57 (0.35-0.95)	3.5 (2.6-5.1)
<i>Dorosoma cepedianum</i>	Omnivorous	0.033 (0.005-0.068)	0.45 (0.10-1.06)	0.26 (0.18-0.39)	0.28 (0.06-0.52)	0.84 (0.19-1.78)	4.0 (2.7-6.3)
<i>Moxostoma macrolepidotum</i>	Omnivorous	0.017 (0.05-0.031)	0.09 (0.07-0.15)	0.18 (0.16-0.20)	0.14 (0.09-0.22)	0.62 (0.43-0.73)	3.3 (2.8-3.6)
<i>Carpoides cyprinus</i>	Omnivorous	0.024 (0.004-0.046)	0.21 (0.02-0.60)	0.17 (0.10-0.30)	0.18 (0.15-0.45)	0.64 (0.09-1.30)	3.4 (2.1-5.5)
<i>Cyprinus carpio</i>	Omnivorous	0.035 (0.011-0.069)	0.16 (0.02-0.46)	0.24 (0.12-0.41)	0.19 (0.04-0.28)	0.56 (0.15-2.13)	10.2 (4.1-16.1)
<i>Esox lucius</i>	Predatory	0.022 (0.13-0.031)	0.13 (0.02-0.22)	0.07 (0.05-0.08)	0.15 (0.08-0.19)	0.34 (0.17-0.61)	2.6 (2.3-2.9)
<i>Micropterus salmoides</i>	Predatory	0.022 (0.004-0.060)	0.11 (0.04-0.24)	0.10 (0.08-0.13)	0.11 (0.05-0.23)	0.59 (0.36-1.13)	3.4 (0.8-5.4)
<i>Morone chrysops</i>	Predatory	0.024 (0.004-0.38)	0.06 (0.04-0.08)	0.19 (0.17-0.24)	0.08 (0.04-0.12)	0.45 (0.18-0.98)	4.5 (2.6-6.3)
<i>Lepisosteus platostomus</i>	Predatory	0.030 (0.004-0.085)	0.14 (0.10-0.20)	0.16 (0.13-0.20)	0.18 (0.07-0.28)	0.74 (0.63-0.85)	3.6 (1.7-7.4)
<i>Micropterus dolomieu</i>	Predatory	0.005 --	0.16 (0.04-0.27)	0.15 (0.14-0.16)	0.13 (0.08-0.19)	0.98 (0.68-1.28)	3.8 (3.5-4.1)
Water (ppb)	--	0.6 (0.1-2)	21 (5-38)	1 (0.1-5)	2 (1-6)	2 (1-18)	31 (1-61)
Sediments	--	2.0 (0.2-12.1)	17 (2-87)	19 (1-82)	27 (3-124)	28 (3-140)	81 (6-335)

\* Adapted from Mathis and Cummings (1973). Values are mean metal concentrations expressed as parts per million dry weight, with ranges shown in parentheses.

Table 15  
Concentrations of Metals in Algae (*Macrocystis purifera*) and in  
Feces of Crabs (*Pugettia producta*) Fed Algae, and Theoretical  
Retention of Metals by the Crabs\*

Metal	Average Concentration, ppm**		Total Theoretical Metal Ingested/ g Feces ( $\mu$ g)†	% Metal Remaining in Feces	% of Ingested Initial Metal Retained Theoretically
	Algae	Feces			
As	77.30 $\pm$ 17.30	147.78 $\pm$ 41.11	463.80	31.9	68.1
Cd	2.27 $\pm$ 0.27	1.78 -	13.62	13.1	86.9
Cr	3.33 $\pm$ 2.33	2.28 $\pm$ 1.83	19.98	11.4	88.6
Cu	4.60 $\pm$ 1.80	16.67 $\pm$ 10.00	27.60	60.4	39.6
Pb	4.33 $\pm$ 1.07	19.44 $\pm$ 3.83	25.98	74.8	25.2
Zn	22.00 $\pm$ 1.47	43.33 $\pm$ 15.56	132.00	32.8	67.2

\* Data adapted from Boothe and Knauer (1972).

\*\* Data were converted from micrograms of metal per gram ash weight by using the authors' conversion factors: data for algae and feces were divided by 1.5 and 1.8, respectively, to convert to dry weight basis.

† Based upon the ratio 6 g algae consumed:1 g feces produced.

Table 16

Heavy Metals in Sediments and Animals from Ocean Sediment-Dumping Sites Along the  
Northeastern Coast of the United States\*

Metal	Sediment	Clam Muscle	Whelk		Crab		Gills	Flounder	
			Muscle	Digestive Gland	Flesh	Digestive Gland		Flesh	Liver
Long Island Sound (Area 2)									
Ag	--	--	<0.2	6.4	0.28	3.1	0.57	<0.1	<0.1
As	--	--	9.0	--	--	--	--	--	--
Cd	--	--	0.17	15.7	1.0	--	2.7	<0.1	--
Cr	--	--	0.8	<0.8	0.6	1.2	2.5	<0.3	<0.5
Cu	--	--	21	1.08	13.0	161	--	1.4	--
Hg	--	--	0.12	0.27	0.18	0.09	--	0.12	--
Pb	--	--	0.07	1.1	3.4	1.3	2.9	<0.6	--
Zn	--	--	29.5	1.03	64.6	31.2	--	5.0	34.1
Long Island (Dumpsite)									
Ag	<1	--	--	--	--	--	--	<0.1	<0.1
As	--	--	--	--	--	--	--	2.7	--
Cd	<1	--	--	--	--	--	--	<0.1	--
Cr	--	--	--	--	--	--	--	0.6	<0.7
Cu	86	--	--	--	--	--	--	0.7	--
Hg	--	--	--	--	--	--	--	0.13	0.22
Pb	49	--	--	--	--	--	--	<0.5	--
Zn	140	--	--	--	--	--	--	5.8	36.8
Chincoteague Inlet (Control Site)									
Ag	--	<0.1	<0.1	6.8	0.26	2.6	0.8	<0.1	<0.1
As	--	1.3	--	--	--	--	--	2.0	--
Cd	--	<0.1	<0.1	7.4	<0.1	4.8	1.1	<0.1	<0.1
Cr	--	<0.5	<0.5	<0.5	<0.5	0.5	0.83	<0.2	<0.3

(Continued)

(Continued)

\* Adapted from Greig et al. (1977). Values are mean metal concentrations expressed as parts per million wet weight.

Table 16 (Concluded)

Metal	Sediment	Clam Muscle	Whelk		Crab		Gills	Flounder	
			Muscle	Digestive Gland	Flesh	Digestive Gland		Flesh	Liver
Chincoteague Inlet (Control Site) (Continued)									
Cu	--	0.9	11.9	32.4	--	66	46.4	--	--
Hg	--	<0.05	0.06	0.16	0.15	0.24	--	0.17	0.13
Pb	--	<0.5	0.9	1.7	<0.5	0.8	0.9	0.8	--
Zn	--	--	15.3	405	35.5	31.5	--	6.3	35.2
Delaware (Dumpsite)									
Ag	--	--	--	--	0.38	3.1	0.7	<0.1	<0.1
As	--	--	--	--	1.9	--	--	1.8	--
Cd	--	--	--	--	0.08	3.2	0.65	<0.1	0.1
Cr	2	--	--	--	<0.3	0.4	0.80	<0.5	<0.5
Cu	4	--	--	--	--	--	26.5	--	10.5
Hg	0.1	--	--	--	0.16	0.07	--	0.15	0.30
Pb	3.6	--	--	--	0.9	1.0	1.5	1.0	1.7
Zn	5.6	--	--	--	37.3	25.9	--	6.2	32.2
New York Bight (Dumpsite)									
Ag	--	--	--	--	0.79	3.4	0.87	<0.1	<0.1
As	--	--	--	--	--	--	--	2.0	--
Cd	--	--	--	--	0.1	1.1	1.0	<0.1	<0.1
Cr	58	--	--	--	<0.5	<0.6	--	<0.2	<0.3
Cu	85	--	--	--	14.8	73	28.6	--	--
Hg	0.2	--	--	--	0.19	1.9	0.03	0.17	0.13
Pb	97	--	--	--	<1.3	1.3	3.1	0.8	--
Zn	154	--	--	--	32.1	18.1	--	6.3	35.2

Table 17  
Arsenic Concentrations (ppm) in Plants, Fish, Fish Stomach  
Contents, and Polychaete Worms from Waterman and Cockburn  
Sound off the Coast of Western Australia\*

Organism	Mean As Level, ppm	Range
<b>Algae</b>		
<i>Ecklonia radiata</i>	11.2	8.1 - 15.9
<b>Seagrasses</b>		
<i>Posidonia australis</i>	--	<0.1 - 0.22
<i>P. ostenfeldi</i>	0.59	--
<i>P. sinuosa</i>	0.51	0.42 - 0.59
<i>Heterozostera tasmanica</i>	0.35	--
<i>Halophila ovalis</i>	0.25	--
<i>Amphibolis griffithii</i>	--	<0.3 - 0.21
<i>A. antarctica</i>	0.19	0.16 - 0.21
<b>Polychaete worms (collected by diving)</b>		
Eunicidae	--	23.0
Spionidae	--	18.6
Aphroditidae	--	7.1 - 10.5
Glyceridae	--	12.7
<b>Gut contents of school whiting (<i>Sillago bassensis</i>)</b>		
Polychaete (unidentified)	--	2.4 - 9.6
Polychaete (Eunicidae)	--	2.4 - 31.3
Polychaete (Orbiniidae)	--	14.6
Echinoderm (Holothuroidea)	--	4.1
Gastropod molluscs	--	1.3 - 2.3
<b>Fish (muscle)</b>		
<i>Sillago bassensis</i> (school whiting)	7.72	3.2 - 14.5
<i>S. maculata</i> (Trumpeter whiting)**	2.37	1.2 - 4.9
<i>Kyphosus sydneyanus</i> (buffalo bream)	--	<0.025 - 0.6

\* Adapted from Edmonds and Francesconi (1981). The authors indicated that fish muscle was analyzed as wet weight, but did not indicate the basis for other organisms. The reviewer, therefore, assumes that all values were given as wet weight.

\*\* *S. maculata* collected at Cockburn Sound; all other species collected at Waterman.

Table 18  
Heavy Metals (ppm) in Tubificid Worms Fed Contaminated  
Bacterial Suspensions\*

<u>Metal</u>	<u>Worms, Initial**</u>	<u>Bacteria</u>	<u>Worms</u>	<u>Bacteria</u>	<u>Worms</u>	<u>Bacteria</u>	<u>Worms</u>
Cr	19.10	109.09	3.92	983.00	14.08	2850.00	29.86
Cu	80.70	213.22	236.17	765.87	397.05	1068.00	621.19
Pb	151.14	119.01	178.95	409.56	559.18	720.80	568.31
Zn	529.80	206.61	261.86	1648.46	685.08	2010.00	868.33

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\* Adapted from Patrick and Loutit (1976). Values are mean metal concentrations expressed as parts per million dry weight.  
 \*\* Field collected and washed.

Table 19  
Concentrations of Six Trace Elements (Individual Values of Ranges,  $\mu\text{g g}^{-1}$  Wet Weight) in Finfish, Bivalve and Cephalopod Molluscs, Crabs, and Shrimps  
or Prawns in Victoria Harbor, Hong Kong\*

Group	Area of Catch	Cd	Cr	Cu	Pb	Hg	Zn
Finfish	Eastern	<0.1	<0.1	0.2	<0.1	<0.1-0.1	2.3-5.5
	Southern	<0.1	<0.1-0.1	<0.1-0.3	<0.1	<0.1-0.1	1.9-6.6
	Harbor approaches east	<0.1	0.1-0.2	<0.2-1.1	<0.1	<0.1	0.8-14.0
	Harbor approaches west	<0.1	<0.1-0.3	<0.1-0.5	<0.1-0.3	<0.1-0.4	6.2-25.4
	Victoria Harbor	<0.1	<0.1	0.1-0.3	<0.1	<0.1-0.1	2.9-3.9
Bivalve molluscs	Southern	0.3-2.6	<0.1	2.1-2.2	0.1-3.0	<0.1	26.2-105.0
	Harbor approaches east	0.25	0.5	5.0	0.5	<0.1	18.8
	Victoria Harbor	<0.1	1.5	6.3	0.4	<0.1	19.4
Cephalopod molluscs	Harbor approaches west	<0.1	<0.1	4.4	<0.1	0.1	15.7
Crabs	Southern	<0.1-2.5	<0.1-0.9	1.1-35.2	<0.1-0.2	<0.1-0.1	10.1-81.6
	Harbor approaches east	<0.1	<0.1	6.4-10.3	<0.1	<0.1	24.2-29.0
	Harbor approaches west	<0.1	<0.1	8.3	<0.1	<0.1	25.6
	Victoria Harbor	0.2	<0.1	10.3	0.3	0.1	17.8
Shrimps or prawns	Eastern	1.3	<0.1	23.4	<0.1	0.1	16.9
	Harbor approaches west	0.3-0.5	<0.1	24.2-28.8	<0.1	<0.1-0.1	13.5-23.9

\* From Phillips et al. (1982). Values are individual metal concentrations or ranges expressed as parts per million wet weight. Published with permission, Copyright 1982, Applied Science Publishers, LTD.



Table 20  
Heavy Metal Concentrations (ppm) in Spring Water, Algae, and  
Microcrustacea in a Laboratory Microcosm\*

Component	Ag	As	Cd	Cr	Cu	Hg	Ni	Pb	Se	Sn	Zn
Water (ppb)	0.0068	0.24	0.016	0.022	3.8	0.13	0.89	0.85	0.03	2.0	12.0
Algae											
Mixed culture	4.6	3.4	1.34	1.2	156.5	4.25	2.7	10.0	1.9	20.0	338.7
<i>Euglena gracilis</i>	4.7	3.6	1.38	1.4	283.2	4.20	1.8	12.6	2.0	23.5	304.0
Microcrustaceans											
<i>Daphnia pulex</i>	0.78	2.3	1.34	1.3	70.0	2.68	4.2	9.8	2.1	4.2	134.6
<i>Daphnia magna</i>	0.46	2.2	1.24	0.6	77.5	3.30	3.6	7.2	1.6	1.2	102.0

\* After Cowgill (1976). Values are mean metal concentrations reported as parts per million dry weight.

Table 21  
Copper and Zinc Concentrations (ppm) in Sediments and Biota  
in Four Desert Streams in Arizona\*

Species	Cu				Zn			
	Pinto Creek	Mine Waste Stream	Camp Creek	Seven Springs Wash	Pinto Creek	Mine Waste Stream	Camp Creek	Seven Springs Wash
Fish ( <i>Angosia chrysogaster</i> )	57 (23-151)	63 (41-87)	15 (6-22)	14 (13-15)	161 (150-176)	53 (31-75)	169 (110-210)	124 (118-130)
Amphibians ( <i>Rana pipiens</i> )	81 (16-221)	--	17 (14-20)	21 (18-26)	248 (165-577)	--	92 (69-114)	73 (72-74)
Insects	187 (143-329)	580 (410-720)	--	39 (21-52)	133 (76-220)	116 (95-130)	--	230 (190-260)
Emergent plants	234 (130-572)	178 (154-202)	66 (57-174)	23 (15-41)	59 (31-116)	69 (40-86)	63 (54-73)	42 (23-69)
Submersed plants/algae	879 (155-1587)	560 (300-710)	--	16 (15-18)	156 (99-216)	71 (55-86)	--	27 (23-30)
Sediments	177 (60-495)	302 (95-510)	23 (21-24)	45 (39-50)	59 (20-112)	80 (56-104)	66 (60-71)	90 (80-101)

\* After Lewis (1980). Copyright, ASTM, 1916 Race Street, Philadelphia, Pa. 19103. Reprinted with Permission. Values are mean metal concentrations, with ranges shown in parentheses, expressed as parts per million dry weight.

Table 22  
Levels of Heavy Metals in Invertebrates and in Muscle Tissue of  
Fish in a Georgia Estuary\*

Species	Primary Foods**	Cd	Cu	Hg	Pb	Zn
<b>Fishes</b>						
<i>Ospanus tau</i>	Decapods	0.06 ± 0.03	1.8 ± 0.8	2.74 ± 1.64	0.16 ± 0.16	45 ± 26
<i>Bairdiella chrysura</i>	Mysids, copepods	0.12 ± 0.12	2.3 ± 1.7	1.07 ± 0.77	0.06 ± 0.05	21 ± 6
<i>Cynoscion regalis</i>	Mysids	0.05 ± 0.02	1.9 ± 0.8	0.44 ± 0.05	0.09 ± 0.05	26 ± 7
<i>Leiostomus xanthurus</i>	Copepods	0.04 ± 0.02	1.8 ± 0.9	0.23 ± 0.13	0.08 ± 0.13	22 ± 13
<i>Micropogon undulatus</i>	Copepods	0.04 ± 0.03	2.3 ± 1.4	0.31 ± 0.22	0.07 ± 0.09	25 ± 19
<i>Stellifer lanceolatus</i>	Copepods	0.04 ± 0.03	1.8 ± 0.8	0.50 ± 0.37	0.13 ± 0.23	26 ± 8
<i>Ancyclopesetia quadrocellata</i>	Mysids	0.22	5.2	0.46	0.08	51
<i>Citharichthys spilopterus</i>	Mysids	0.07	1.2	0.17	--	24
<i>Ftropus crosotus</i>	Copepods	0.07	0.9	0.10	0.01	21
<i>Scophthalmus aquosus</i>	Mysids	0.02	2.0	1.03	0.01	31
<i>Symphurus plagiosa</i>		0.03 ± 0.02	1.6 ± 1.0	0.63 ± 0.58	0.09 ± 0.16	21 ± 8
<b>Crustaceans</b>						
<b>Decapods</b>						
<i>Palaemonetes pugio</i>	Detritus, scavengers	0.06	74	0.13	0.04	58
<i>Trachypeneus constrictus</i>	Detritus, scavengers	0.11	56	--	0.03	55
<i>Xiphopeneus kroyeri</i>	Detritus, scavengers	0.36 ± 0.56	22 ± 6	0.26 ± 0.02	0.06 ± 0.02	60 ± 7
<i>Panaeus aztecus</i>	Detritus, scavengers	0.07 ± 0.01	18 ± 5	0.17 ± 0.07	0.20 ± 0.23	56 ± 5
<i>Panaeus setiferous</i>	Detritus, scavengers	0.04 ± 0.03	16 ± 4	0.23 ± 0.15	0.12 ± 0.20	53 ± 14
<i>Hexapanopeus augustifrons</i>	Detritus, scavengers	0.06	13	--	0.07	58
<i>Menippe mercenaria</i>	Detritus, scavengers	0.13	43	1.57	--	290

(Continued)

- \* After Stickney et al. (1975). Values are means ± one standard deviation, expressed as parts per million dry weight.
- \*\* For fish, this is based upon gut contents and includes groups which represent 20% of the food of the fish; for invertebrates this is based upon general feeding habits.

Table 22 (Concluded)

Species	Primary Foods	Cd	Cu	Hg	Pb	Zn
<i>Callinectes sapidus</i> (muscle)	Detritus, scavengers	0.07 ± 0.08	31 ± 12	0.68 ± 0.33	0.05 ± 0.05	185 ± 95
(eggs)	Detritus, scavengers	0.08 ± 0.03	41 ± 14	0.20 ± 0.11	0.94 ± 1.30	160 ± 26
(gills)	Detritus, scavengers	0.22 ± 0.39	60 ± 44	0.18 ± 0.04	0.84 ± 1.09	92 ± 21
Amphipods						
<i>Gammarus</i> sp.	Plankton	0.01	36	--	--	70
Copepods						
<i>Pseudodiaptomus coronatus</i> and <i>Acartia tonsa</i> (mixed)	Microplankton	0.01	19	--	0.23	100
Mysids						
<i>Neomysis americana</i>	Plankton	0.40 ± 0.52	25 ± 11	0.06 ± 0.06	0.03 ± 0.03	87 ± 30

Table 23

## Heavy Metals in Omnivorous and Carnivorous Fishes and Their Prey in the Loire Estuary\*

Fish Species	Prey Species	Cd		Cu		Pb		Zn	
		Fish	Prey	Fish	Prey	Fish	Prey	Fish	Prey
Banc de Bilho									
Omnivorous species									
<i>Dicentrarchus labrax</i>	<i>Neomys integer</i> (mysid)	0.116	0.069	3.306	20.492	0.584	4.652	81.5	47.8
	<i>Crangon crangon</i> (crab)	0.074	0.140	3.502	38.638	0.606	2.703	79.0	46.1
<i>Gobius microps</i>	<i>C. crangon</i>	0.024	0.660	1.025	36.415	0.977	4.325	68.5	44.6
	<i>Boccardia ligerica</i> (annelid)	0.102	4.027	2.031	12.898	1.543	16.205	82.6	245.1
	<i>N. integer</i> and <i>C. crangon</i>	0.005	0.026	2.332	35.257	1.416	4.046	68.2	64.1
	<i>C. crangon</i>		0.094		8.416		2.874		36.5
	<i>N. integer</i> and <i>C. crangon</i>	0.044	0.069	1.203	20.492	0.360	4.652	58.0	47.8
	<i>C. crangon</i>	0.005	0.102		22.425		2.520		39.0
	<i>C. crangon</i>	0.005	0.109	1.025	37.124	2.055	4.844	67.6	47.3
Carnivorous species									
<i>Merlangius merlangus</i>	<i>C. crangon</i> and <i>N. integer</i>	0.005	0.094	2.945	8.416	1.741	2.874	32.9	36.5
	<i>N. integer</i>		0.026		32.257		4.046		64.1
		0.037	0.069	1.261	20.492	0.693	4.652	50.1	47.8
<i>Gadus luscus</i>	<i>C. crangon</i>	0.015	0.183	2.882	5.970	1.796	4.256	29.5	36.4
Banc de Pipy									
Omnivorous species									
<i>Dicentrarchus labrax</i>	Zooplankton	0.106	1.246	4.167	31.291	0.416	19.498	68.9	288.3
	Zooplankton	0.048	0.707	4.853	17.753	0.656	61.611	101.3	447.9
<i>Gobius microps</i>	<i>C. crangon</i>	0.023	0.403	1.511	29.903	0.671	3.515	79.1	41.7
	Zooplankton	0.016	1.146	1.704	31.291	0.726	19.498	81.6	288.3
(Continued)									

(Continued)

\* After Metayer et al. (1980). Data are means expressed as parts per million dry weight.

Table 23 (Concluded)

Fish Species	Prey Species	Cd		Cu		Pb		Zn	
		Fish	Prey	Fish	Prey	Fish	Prey	Fish	Prey
Carnivorous species									
<i>Merlangius merlangus</i>	<i>C. crangon</i> ,	0.008	0.113	1.228	34.000	1.248	5.507	38.2	40.4
	<i>Mesopodopsis slabberi</i> ,		0.098		7.257		12.054		134.0
	<i>G. microps</i> , and		0.016		1.802		1.685		56.3
	<i>Osmerus eperlanus</i> (fish)		0.006		2.477		4.225		71.1
<i>Stizostedion lucioperca</i>	<i>G. microps</i> ,	0.093	0.074	3.093	2.139	0.573	1.450	75.7	72.6
	<i>O. eperlanus</i> ,		0.024		3.465		1.885		74.6
	and <i>C. crangon</i>		0.966		33.472		3.215		51.5

Table 24

A Comparison of the Uptake of Methylmercury from Food and Water  
by *Daphnia pulex* and *Salmo gairdneri*\*

Source of Methylmercury	Exposure Level (as Hg) (ppb)	Accumulation (ppm as Hg) of Methylmercury at Differing Lengths of Exposure											
		<i>Daphnia pulex</i> , hours						<i>Salmo gairdneri</i> , weeks					
		1	4	8	24	48	72	1	2	3	6	12	
Water	0	0	0	0	0	0	0	0	0	0	0	0	0
	0.25	2.5	2	4	4	5	3	0.7	1.0	1.1	1.4	1.5	
	0.5	6	6	7	7	7.5	7	1.5	2.0	2.6	2.7	4.0	
	1.0	9	11.5	11	10	11	10.5	2.6	3.4	3.6	4.5	5.4	
	2.5	17.5	18	18.5	16	16	17	4.4	5.2	5.8	5.7	6.8	
	5.0	22	21.5	21	23	21	21	5.8	8.3	9.5	10.1	10.3	
<i>Chlamydomonas</i>	(ppm)												
	0	0	0	0	0	0	0						
	9	0.2	0.3	0.6	0.5	0.6	0.7						
	16	0.9	0.8	0.8	0.9	1.0	0.8						
	39	1.1	1.2	1.2	1.8	1.6	1.9						
	74	2.1	2.0	2.2	2.5	2.6	2.5						
Diet treated with methylmercury	126	2.8	2.9	3.3	3.8	3.7	4.0						
	0							0	0	0	0	0	0
	3.4							0.6	1.1	1.2	1.3	2.3	
	6.2							1.0	1.6	1.9	2.5	3.6	
	9.3							1.3	2.1	2.2	2.7	4.2	
	15.8							1.8	2.7	3.4	4.0	4.5	
	21.6							2.2	3.2	3.3	4.4	5.1	

\* After Lock (1975). Data expressed on the basis of parts per million dry weight for *Daphnia* and parts per million wet weight for *Salmo*. All data were converted from the author's graphs.

Table 25

## Methylmercury Exposure and Uptake of Methylmercury by Rainbow Trout\*

Methylmercury Concentration in Water, $\mu\text{g Hg}/\ell^{**}$	Methylmercury Consumption Rate, ng Hg/g fish per day	Body Burden of Mercury, † $\mu\text{g Hg/g}$ Wet Fish			Methylmercury Accumulation Rate, $\mu\text{g Hg/g/day}$
		8 days	15 days	24 days	
Control	139	0.87, 0.99 (2)	1.78, 1.93 (2)	2.06 $\pm$ 0.43 (4)	0.10
Control	265	1.41, 1.59 (2)	2.62, 2.88 (2)	3.41 $\pm$ 0.35 (4)	0.16
Control	379	1.83, 2.78 (2)	3.63, 4.20 (2)	5.67 $\pm$ 1.03 (4)	0.25
0.33 $\pm$ 0.04	0	0.27 $\pm$ 0.04 (6)	0.38 $\pm$ 0.05 (6)	0.76 $\pm$ 0.16 (11)	0.03
0.33 $\pm$ 0.04	139	1.11, 1.13 (2)	1.62, 3.55 (2)	3.30 $\pm$ 0.14 (3)	0.14
0.33 $\pm$ 0.04	265	1.73 (1)	2.91, 3.17 (2)	4.08 $\pm$ 0.50 (4)	0.18
0.33 $\pm$ 0.04	379	1.55, 2.68 (2)	3.85, 4.87 (2)	5.38 $\pm$ 0.44 (4)	0.25
1.33 $\pm$ 0.16	0	1.29 $\pm$ 0.17 (6)	1.86 $\pm$ 0.16 (6)	2.70 $\pm$ 0.48 (12)	0.13
1.33 $\pm$ 0.16	139	1.11, 2.36 (2)	3.44 (1)	4.95 $\pm$ 0.71 (4)	0.22
1.33 $\pm$ 0.16	265	2.83, 3.63 (2)	4.22, 5.28 (2)	6.53 $\pm$ 0.84 (4)	0.31
1.33 $\pm$ 0.16	379	3.65, 4.56 (2)	5.42, 7.09 (2)	8.63 $\pm$ 0.59 (4)	0.41

\* After Phillips and Buhler (1978).

\*\* Values for Hg in water were 0.35 and 1.38  $\mu\text{g}/\ell$  for the rate study.† Values shown are means  $\pm$  one standard deviation, with the sample size shown in parentheses.



Table 26  
Mercury Levels (ppm) in Fish and Benthic Invertebrates  
from the North Fork Holston River\*

<u>Year</u>	<u>Station</u>	<u>Total Hg in Benthic Invertebrates</u>	<u>Total Hg in Rock Bass</u>	<u>Total Hg in Hog Sucker</u>
1974	McCrary	0.051	0.493	0.409
	Below ponds	1.550	1.623	1.420
	Hayters	0.790	1.623	1.235
	Kingsport	0.201	0.656	0.427
1975	Below ponds	2.028		2.140
	Hayters	0.556		1.661
	Kingsport	0.217	0.921	0.371
	Chatham	0.052	0.213	0.198
	Neil	0.043	0.322	0.223
	McKenna	0.876	1.814	2.066
	Hines	0.416	1.290	1.110

\* After Hildebrand, Strand, and Huckabee (1980). Reprinted with permission from the Journal of Environmental Quality, Vol 9, p 397, Copyright 1980--American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America. Values are mean metal concentrations expressed as parts per million wet weight.

Table 27  
Mercury in Fish From Northwestern Ontario\*

Locality/Species	N	Mercury Residues, ppm		Body Weight, g	
		$\bar{X}$	Range	$\bar{X}$	Range
Clay Lake					
Burbot	4	21.95	19.10-24.80	1,692	1,355-2,150
Walleye	5	15.74	12.30-19.60	1,350	1,025-2,040
Sucker	5	3.13	2.29-3.75	710	575-1,050
Wabigoon River					
Northern pike	3	15.17	8.57-27.80	1,560	510-2,700
Walleye	4	6.80	0.50-10.40	954	160-1,490
Sucker	3	4.19	0.64-8.94	745	570-1,060
Ball Lake					
Northern pike	7	7.73	1.61-20.00	2,016	1,520-2,900
Rock bass	9	6.22	1.14-10.90	1,083	650-1,685
Walleye	9	4.51	0.52-8.74	1,069	410-1,550
Tide Lake					
Northern pike	4	3.74	2.91-4.88	1,908	1,675-2,365
Walleye	10	3.39	0.28-12.70	2,563	1,530-4,220
Indian Lake					
Walleye	5	2.71	2.39-3.10	1,071	715-1,250
Grassey Narrows Lake					
Walleye	3	2.79	2.75-2.84	1,110	725-1,680
Tetu Lake					
Northern Pike	1	5.25		2,640	
Walleye	1	1.50		1,190	
Sucker	1	1.00		800	
Rock bass	1	0.55		1,800	
Wagigoon Lake					
Northern pike	4	1.37	0.94-1.88	2,232	1,570-3,160
Walleye	7	0.78	0.60-1.10	1,245	870-2,330

\* After Fimreite and Reynolds (1973). Reprinted with permission from the Journal of Wildlife Management, Vol 37, pp 64, 65, Copyright 1973--The Wildlife Society. Metal concentrations are expressed as parts per million wet weight of lateral musculature.

Table 28  
Mercury Residues in Lateral Muscle of Fish from Canadian Waters  
Where Mercury Contamination was Suspected\*

Locality/Species	Mercury Residues, ppm			Body Weight, g	
	N	$\bar{X}$	Range	$\bar{X}$	Range
PINCHI LAKE, B. C.					
<i>Salvelinus namaycush</i> (lake trout)	2	5.78	1.07-10.5	1700	1700-1700
<i>Mylocheilus caurinus</i> (peamouth)	1	0.84		50	
<i>Prosopium williamsoni</i> (mountain whitefish)	4	0.65	0.30-1.50	307	230-429
<i>Salmo gairdnerii</i> (rainbow trout)	4	0.38	0.25-0.68	243	161-322
LAKE HURON, ONT., south end					
<i>Stizostedion v. vitreum</i> (walleye)	8	1.08	0.58-2.74	807	725-984
ST. CLAIR RIVER, ONT.					
<i>Ambloplites rupestris</i> (rock bass)	6	2.80	0.55-4.64	646	55-368
<i>Lepomis gibbosus</i> (pumpkinseed)	3	2.64	0.26-7.09	64	46-95
<i>Morone chrysops</i> (white bass)	1	1.62		75	
<i>Stizostedion v. vitreum</i> (walleye)	6	1.60	0.89-2.43	646	370-1018
<i>Esox lucius</i> (northern pike)	1	1.00		2265	
LAKE ST. CLAIR, ONT.					
<i>Stizostedion v. vitreum</i> (walleye)	8	2.88	1.29-5.01	819	363-1928
LAKE ERIE, ONT., west end					
<i>Stizostedion v. vitreum</i> (walleye)	8	0.71	0.58-0.90	595	462-907
OTTAWA RIVER, ONT., downstream from pulp mill					
<i>Stizostedion canadense</i> (sauger)	10	1.48	0.47-2.73	144	23-389
OTTAWA RIVER, ONT., upstream from pulp mill					
<i>Stizostedion canadense</i> (sauger)	10	0.72	0.42-1.00	165	117-217
ST. MAURICE RIVER, QUE., downstream from chlorine plant					
<i>Stizostedion v. vitreum</i> (walleye)	4	2.09	1.96-2.15	390	312-482
<i>Catostomus catostomus</i> (longnose sucker)	1	0.88		397	
<i>Semotilus corporalis</i> (fallfish)	2	0.84	0.73-0.94	128	114-142
<i>Esox lucius</i> (northern pike)	1	0.75		312	
<i>Catostomus commersonii</i> (white sucker)	4	0.73	0.52-0.95	118	4-454
<i>Perca flavescens</i> (yellow perch)	4	0.65	0.26-0.82	49	2-142

(Continued)

\* After Fimreite et al. (1971). Reprinted with permission from Canadian Field-Naturalist, Vol 85, pp 213, 214, Copyright 1971--Ottawa Field-Naturalists' Club. Values are expressed as parts per million wet weight.

Table 28 (Concluded)

Locality/Species	Mercury Residues, ppm			Body Weight, g	
	N	$\bar{X}$	Range	$\bar{X}$	Range
ST. MAURICE RIVER, QUE., upstream from chlorine plant					
<i>Stizostedion v. vitreum</i> (walleye)	18	0.69	0.48-1.20	487	142-1988
<i>Esox lucius</i> (northern pike)	5	0.42	0.30-0.73	494	198-1448
<i>Perca flavescens</i> (yellow perch)	2	0.20	0.19-0.20	1.5	1-2
<i>Culaea inconstans</i> (brook stickleback)	2	0.19	0.19-0.20	1	1-1
PORT ALBERNI, B. C.					
<i>Sebastes caurinus</i> (copper rockfish)	4	0.60	0.07-1.13	636	332-870
<i>Ophiodon elongatus</i> (lingcod)	2	0.26	0.24-0.27	823	789-857
NANAIMO, B. C.					
<i>Sebastes caurinus</i> (copper rockfish)	4	0.37	0.26-0.48	1130	765-1656
<i>Ophiodon elongatus</i> (lingcod)	1	0.08		871	
HORSESHOE BAY, B. C.					
<i>Sebastes caurinus</i> (copper rockfish)	1	0.18		353	
<i>Ophiodon elongatus</i> (lingcod)	1	0.08		610	
BAIE DES CHALEURS (Bathurst, N. B.)					
<i>Pseudopleuronectes americanus</i> (winter flounder)	2*	1.10	0.86-1.33	215	
<i>Anguilla rostrata</i> (American eel)	4	0.32	0.28-0.38	205	129-324
<i>Microgadus tomcod</i> (Atlantic tomcod)	1*	0.18		100	
<i>Alosa pseudoharengus</i> (alewife)	2*	0.10	0.10-0.10	81	
BAIE DES CHALEURS (Dalhousie, N. B.)					
<i>Clupea h. harengus</i> (Atlantic herring)	4	0.04	0.03-0.06	236	186-288

\* *P. americanus* and *A. pseudoharengus* - two analyses of two pooled samples each containing four fish. *M. tomcod* - one analysis of a pooled sample of four fish.

Table 29  
Mercury Concentrations in Fishes from Three Northern Maine Lakes\*

<u>Lake and Species**</u>	<u>Total Length cm</u>	<u>Mercury Concentration ppm</u>
Eagle Lake		
Lake Trout	18 - 69	0.13 - 1.11
Brook Trout	28 - 34	0.13 - 0.23
Lake Whitefish	25 - 51	0.30 - 2.17
Burbot	43 - 64	0.40 - 1.29
St. Froid Lake		
Lake Trout	40 - 59	0.34 - 0.84
Brook Trout	24 - 43	0.08 - 0.58
Burbot	32 - 52	0.35 - 0.89
Rainbow Smelt	18 - 22	0.28 - 0.59
Cliff Lake		
Lake Trout	35 - 47	0.10 - 0.23
Brook Trout	33 - 38	0.12 - 0.21

\* From Akielaszek and Haines (1981). Reprinted with permission from the Bulletin of Environmental Contamination and Toxicology, Vol 27, p 203, Copyright 1981--Springer-Verlag. Metal concentrations are expressed as parts per million wet weight of muscle.

\*\* Rainbow smelt are forage species for whitefish, lake trout, and burbot. The smelt were present in Eagle and St. Froid Lakes but were absent at Cliff Lake.

Table 30

Mean Concentrations of  $^{203}\text{Hg}$  Accumulated in Experimental Oyster Tissues\*

Oyster tissue	$^{203}\text{Hg}$ , $\mu\text{g/kg}$ wet tissue $\pm 95\%$ confidence limits**		
	Control	Mercury reducer	Mercury accumulator
Mantle fluid	3.5 $\pm$ 2.2	9.7 $\pm$ 8.0	11.8 $\pm$ 7.3
Mantle	199.9 $\pm$ 173.0	511.4 $\pm$ 357.8	405.4 $\pm$ 215.0
Gills	647 $\pm$ 538.7	<u>1747.3 <math>\pm</math> 880.7</u>	<u>2849.9 <math>\pm</math> 1282.0</u>
Viscera	216.4 $\pm$ 177.7	463.2 $\pm$ 393.0	335.7 $\pm$ 210.2
Adductor muscle	56.1 $\pm$ 42.5	133.0 $\pm$ 129.3	<u>161.2 <math>\pm</math> 78.5</u>
Whole oyster	201.4 $\pm$ 163.5	312.4 $\pm$ 238.0	<u>463.0 <math>\pm</math> 258.4</u>

\* From Sayler, Nelson, and Colwell (1975). Published with permission, Copyright 1975, American Society for Microbiology.

\*\* Average of eight oysters. Underlined values are significantly different from controls.

Table 31  
Mercury Concentrations in Fish and Invertebrates in  
North American Offshore Waters\*

	No. Col- lections Analyzed**	Mercury Content, ppm wet weight				Whole Animal
		Muscle		Liver		
		Range	Average	Range	Average	
Fish						
Bottom feeders						
American dab	2	0.06-0.08	0.07	0.11-0.14	0.13	
Atlantic cod	2	0.14-0.25	0.20	0.11-0.20	0.16	
Atlantic wolf- fish	2	<0.05-0.15	0.08	<0.05-0.06	<0.05	
Blackbelly rosefish	1	0.22	0.22	0.40	0.40	
Black sea bass	1	0.08	0.08	0.18	0.18	
Cusk	4	0.15-0.49	0.31	0.14-0.83	0.42	
Fourspot flounder	2	0.16	0.16	0.23-0.27	0.25	
Gulf Stream flounder	2	0.05	0.05	ND†	ND	
Haddock	2	0.05-0.09	0.06	<0.05	<0.05	
Little skate	2	0.13-0.16	0.15	0.10-0.23	0.17	
Longhorn sculpin	2	0.08-0.09	0.09	0.09-0.16	0.13	
Ocean pout	2	<0.05-0.11	0.07	<0.05-0.09	0.06	
Red hake	2	<0.05-0.05	<0.05	<0.05-0.08	0.06	
Striped sea- robin	1	0.35	0.35	0.38	0.38	
Thorny skate	2	0.21-0.26	0.24	0.09-0.15	0.12	
White hake	2	0.10-0.12	0.11	0.12-0.16	0.14	
Windowpane flounder	1	0.10	0.10	0.12	0.12	
Winter flounder	3	0.06-0.14	0.09	0.07-0.18	0.11	
Winter skate	1	0.15	0.15	0.18	0.18	
Witch flounder	2	0.07-0.10	0.09	0.13-0.16	0.15	
Yellowtail flounder	2	0.10-0.24	0.17	0.17-0.25	0.21	
Pelagic feeders						
Pollock	2	0.08-0.10	0.09	<0.05-0.06	<0.06	
Redfish	2	0.10-0.20	0.15	0.15	0.15	
Spot	1	<0.05	<0.05	<0.05	<0.05	

(Continued)

\* After Greig, Wenzloff, and Shelpuk (1975).

\*\* Each collection includes 610 animals.

† ND = no data.

Table 31 (Continued)

	No. Col- lections Analyzed	Mercury Content, ppm wet weight				Whole Animal
		Muscle		Liver		
		Range	Average	Range	Average	
Silver hake	1	0.09	0.09	0.10	0.10	
Plankton feeders						
American Shad	1	0.05	0.05	0.67	0.67	
Atlantic herring	2	<0.05-0.09	0.06	0.26-0.28	0.27	
Mackerel	1	0.08	0.08	ND	ND	
Miscellaneous						
Angel shark	1	0.08	0.08	<0.05	<0.05	
Cuskeel	1	0.11	0.11	0.19	0.12	
Spiny dogfish	8	0.07-0.53	0.32	<0.05-0.19	0.10	
Invertebrates						
Lobster		0.31*		0.60		
Pandallid shrimp						0.09
						<0.05
						<0.05
Scallops		<0.05				
Squid						<0.05
						<0.05
						<0.05
						0.06

\* Lobster muscle sample from tail only.



Table 32  
Mercury Concentrations in Edible Finfish from the Derwent Estuary, Tasmania\*

Feeding Habit/Species	Common Name	Number in sample	No. of Fish Having Given Hg Concn. or in Given Range, Wet Weight															
			0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0	1.1-1.5	1.6-2.0				
Vertebrate Predators																		
<i>Galeorhinus australis</i>	School shark	6	1		1		2								1		1	
<i>Squalus</i> sp.	Dog shark	7				2			1	1					2		1	
<i>Leionura atun</i>	Snoek	10		2	2	2	3	1										
<i>Platycephalus bassensis</i>	Sand flathead	25	1	2	4	4	5	3	1						3		2	
<i>Arripis trutta</i>	Australian salmon	27	4	4	4	4	7	4										
Subtotals			6	8	11	12	17	9	2			3			3		4	
Invertebrate Predators																		
<i>Callorhynchus milii</i>	Elephant shark	18		1	6	3	2	1	1	3					1			
<i>Mustelus antarcticus</i>	Gummy shark	8				1	1		2	3					1			
<i>Seriola maculata</i>	Mackerel trevally	12	3	5	1	2		1										
<i>Physiculus barbatus</i>	Cod	7		2	3	2												
<i>Pseudolabrus</i> sp.	Parrot fish	6		1		1	2	1		1								
<i>Latridopsis forsteri</i>	Trumpeter	47	10	13	18	4	1											
<i>Nemadactylus macropterus</i>	Perch	16	3	5	3	2	1			1	1							
<i>Ammotretis</i> sp.	Flounder	8	1	4	1	2												
<i>Sillaginodes punctatus</i>	Spotted whiting	5					1	2	1						1			
Subtotals			17	31	32	17	8	6	4	8	1	1			2			
Herbivores																		
<i>Mugil cephalus</i>	Sea mullet	38	19	11	6		1	1										
<i>Navodon</i> sp.	Leatherjackets	18	4	6	2	4	1	1										
Subtotals			23	17	8	4	2	2										
Grand totals			46	56	51	33	27	17	6	8	1	4			5		4	

\* After Ratowsky, Dix, and Wilson (1975). Reprinted with permission from Australian Journal of Marine and Freshwater Research, Vol 26, p 227, Copyright 1975--C.S.I.R.O.

Table 33  
Concentrations (ppm Dry Weight) of Total Mercury and Methylmercury in Tissues  
of Aquatic Animals from the Southeastern United States\*

Organism and Tissue Examined	No. of Samples	Total Mercury		Methylmercury	
		Av.	Range	Av.	% of Total Hg
<b>Osteichthyes</b>					
Anguillidae, freshwater eels					
<i>Anguilla rostrata</i> (American eel) muscle	1	1.2		0.85	71
Clupeidae, herrings					
<i>Brevoortia tyrannus</i> (Atlantic menhaden) muscle	2	0.3	0.14-0.47	0.25	0.21-0.29
Ariidae, sea catfishes					
<i>Arius felis</i> (sea catfish) muscle	3	1.8	1.5-2.2	1.41	1.1-1.6
Batrachoididae, toadfishes					
<i>Opsanus tau</i> (oyster toadfish) muscle	1	2.9		1.4	48
Cyprinodontidae, killifishes					
<i>Fundulus heteroclitus</i> (mummichog) muscle	1	0.47		0.14	30
Serranidae, sea basses					
<i>Centropristis striata</i> (black sea bass) muscle	2	0.48	0.49-1.2	0.47	0.21-0.72
Rachycentridae, cobias					
<i>Rachycentron canadus</i> (cobia) muscle	1	0.93		0.70	75
Sparidae, porgies					
<i>Archosargus probatocephalus</i> (sheepshead) muscle	1	0.75		0.46	61
Sciaenidae, drums					
<i>Micropogon undulatus</i> (Atlantic croaker) muscle	6	0.38	0.20-0.84	0.25	0.09-0.50
<i>Bairdiella chrysura</i> (silver perch) muscle	2	0.46	0.32-0.60	0.39	0.33-0.46
<i>Cynoscion regalis</i> (weakfish) muscle	1	0.47		0.23	49
<i>Cynoscion nebulosus</i> (spotted sea trout) muscle	1	0.73		0.73	100

(Continued)

\* After Gardner et al. (1975).

Table 33 (Continued)

Organism and Tissue Examined	No. of Samples	Total Mercury		Methylmercury		% of Total Hg
		Av.	Range	Av.	Range	
<i>Leiostomus xanthurus</i> (spot) muscle	5	0.25	0.16-0.38	0.20	0.08-0.27	80
<i>Menticirrhus americanus</i> (southern kingfish) muscle	1	0.36		0.13		36
<i>Stellifer lanceolatus</i> (star drum) muscle	2	0.65	0.50-0.79	0.35	0.27-0.42	54
Bothidae, lefteye flounder						
<i>Paralichthys lethostigma</i> (southern flounder) Muscle	6	0.64	0.39-0.92	0.36	0.14-0.54	56
Liver	1	0.10		0.05		50
<i>Ancyllopsetta quadrocellata</i> (ocellated flounder) muscle	1	0.46		0.41		89
<i>Etropus crossotus</i> (fringed flounder) muscle	1	0.10		ND*		--
<i>Citharichthys spilopterus</i> (bay whiff) muscle	1	0.17		0.16		94
Cynoglossidae, tonguefishes						
<i>Symphurus plagiusa</i> (blackcheek tonguefish) muscle	1	0.10		0.16		100
Chondrichthyes						
Carcharhinidae, requiem sharks						
<i>Megapristis brevirostris</i> (lemon shark) Muscle	4	3.8	3.2-4.6	3.1	3.0-3.1	82
Spleen	6	10.4	1.1-1.5	0.17	ND*-0.55	2
Liver	5	5.5	3.9-8.3	0.05	ND*-0.07	1
<i>Carcharhinus leucas</i> (bull shark) Muscle	1	10.2		3.9		38
Spleen	1	4.6		0.24		5
<i>Carcharhinus limbatus</i> (blacktip shark) Muscle	2	0.75	0.66-0.85	0.52	0.40-0.65	69
Liver	1	1.30		0.06		5

(Continued)

\* ND = not detectable.

(Sheet 2 of 3)

Table 33 (Concluded)

Organism and Tissue Examined	No. of Samples	Total Mercury		Methylmercury		% of Total Hg
		Av.	Range	Av.	Range	
<i>Carcharhinus falciformis</i> (silky shark)						
Muscle	1	1.79		0.74		41
Liver	1	0.75		0.03		4
Squalidae, dogfish sharks						
<i>Squalus acanthias</i> (spiny dogfish) muscle	1	4.0		3.0		75
Sphyrnidae, hammerhead sharks						
<i>Sphyrna lewini</i> (scalloped hammerhead) muscle	1	3.4		0.72		21
Dasyatidae, stingrays						
<i>Dasyatis sabina</i> (Atlantic stingray) muscle	4	2.6	0.30-3.8	2.4	0.24-3.4	92
<i>Dasyatis sayi</i> (bluntnose stingray) muscle	2	1.5	1.3-1.7	0.78	0.31-1.2	52
Crustacea						
Portunidae, crab						
<i>Callinectes sapidus</i> (blue crab) muscle	8	0.45	0.50-1.1	0.31	0.15-0.57	69
Penaeidae, shrimp						
<i>Penaeus setiferus</i> (white shrimp) muscle	9	0.31	0.04-0.65	0.17	0.01-0.47	55
<i>Penaeus aztecus</i> (brown shrimp) muscle	10	0.17	0.07-0.26	0.08	0.01-0.24	47
<i>Xiphopenaeus kroyeri</i> (shrimp) muscle	3	0.26	0.24-0.28	0.06	0.01-0.09	23
Palaemonidae						
<i>Palaemonetes pugio</i>	1	0.13		0.06		46

Table 34  
Concentrations of Total Mercury in Marine Animals\*

Sample Name	Body Length		Total Hg** ppm dry weight	Food Habit
	cm			
Tuna				
Skipjack, <i>Katsuwonus pelamis</i>		1.5±0.4		Fish and squid
Yellowtail, <i>Seriola quinqueradiata</i>		0.85±0.23 0.19±0.14		
Pacific ocean perch, <i>Sebastes alutus</i>	30-40	0.60±0.21		Shrimp, polychaeta and euphausiids living in depth below 300 m
Japanese rockfish				
<i>Sebastes baramenue</i>	50-55	0.62±0.26		
<i>Sebastes flammus</i>	40-50	1.4±0.4		
<i>Sebastes iracundus</i>	40-70	1.6±0.4		
Channel rockfish, <i>Sebastes macrochir</i>	30-35	0.36±0.1		
Common sea bass, <i>Lateolabrax japonicus</i>	60-65	0.91±0.26		Shrimp, shell, euphausiids, copepods and polychaeta in shallow layer
Flounder, <i>Limanda herzenstein</i>	23-30	0.21±0.05		
Black rockfish, <i>Sebastes inermis</i>	20-30	0.33±0.12		
Red sea bream snapper, <i>Pagrus major</i>	23-26	0.25±0.01		
Squid, <i>Todarodes pacificus</i>		0.08±0.02		
Red salmon, <i>Onchorhynchus nerka</i>	55-60	0.15±0.01		Euphausiids and other zooplankton
Alask pollock, <i>Theragra chalcogramma</i>	53-57	0.16±0.13		phytoplankton
Pilchard, <i>Sardinops melanosticta</i>				
<i>Engraulis japonica</i>	10-15	0.14±0.05		
Saury, <i>Cololabis saira</i>	25-30	0.13±0.02		
Surf clam, <i>Spisula (Pseudocardium) sachalinensis</i>		0.07±0.02		Diatoms and dead organisms
Scallop, <i>Patinopecten (Mizuhopecten) yessoensis</i>		0.04±0.01		

\* After Matsunaga (1978).

\*\* The values shown are assumed by the reviewer to be means ± standard deviations.

Table 35  
Mercury Concentrations and Body Weights in Pelagic Marine  
Organisms from the Mediterranean and Atlantic\*

Species	No. of Specimens	Hg Concentration µg/kg fresh weight		Body weight, g		Ocean
		$\bar{X}$	Range	$\bar{X}$	Range	
					56000-79000	
<i>Belone belone</i>	2	210	165-270	173.7	158.0-190.0	Med.
<i>Loligo vulgaris</i>	10	320	85-530	96.7	17.2-244.7	Atl.
	10	100	85-130	74.4	45.4-124.1	North Sea
<i>Sardina pilchardus</i>	26	220	110-330	35.4	26.9-50.4	Med.
	9	300	160-475	47.1	42.0-57.0	Med.
	28	30	5-50	27.3	17.3-37.2	Atl.
<i>Engraulis encrasicolus</i>	20	105	70-140	16.1	13.2-21.0	Med.
	16	380	210-590	39.5	34.3-47.2	Med.
	10	70	50-95	17.6	14.2-20.4	Atl.
<i>Scomber japonicus</i>	9	75	45-95	71.1	50.1-133.6	Atl.
<i>Scomber scombrus</i>	15	340	130-510	87.0	33.1-111.9	Med.
	10	80	45-125	265.6	212.8-302.2	North Sea
	5	330	160-550	312.3	234.5-372.0	North Sea
	3	420	180-550	588.3	500.0-705.0	North Sea
<i>Trachurus trachurus</i>	5	1255	645-2210	225.0	170.4-299.0	Med.
	3	725	315-1400	333.3	255.0-485.0	North Sea
<i>Sarda sarda</i>	10	460	180-815	1355.0	1060-1980	Atl.
<i>Xiphias gladius</i>	2	1260	620-1790	67500	56000-79000	Med.

\* After Stoeppler et al. (1979). Reprinted with permission from Science of the Total Environment, Vol 13, p 214, Copyright 1979--Elsevier Scientific Publishing Co.

Table 36  
Retention of Ingested PCB's by Rainbow Trout\*

<u>Time, Weeks</u>	<u>Ave. <math>\mu</math>g of PCB Consumed/Fish/ 4-Week Period</u>	<u>Total <math>\mu</math>g of PCB Accum./Fish</u>	<u>% Retention of Total PCB Ingested</u>
	11	3	27
8	23	15	48
12	32	26	41
16	45	55	51
20	99	133	64
24	191	276	69
28	197	404	68
32	343	636	68
Total Ingested	938		

\* From Lieb, Bills, and Sinnhuber (1974). Reprinted with permission from the Journal of Agricultural and Food Chemistry, Vol 22, p 641. Copyright 1974--American Chemical Society.

Table 37  
Effect of Duration of Exposure to Diets on the PCB Concentration  
of Yellow Perch Fillets\*

<u>Duration of Exposure, Weeks</u>	<u>PCB Concentration of Fillets, ppm**</u>	
	<u>W-3 Experimental Diet</u>	<u>Control Diet</u>
0	0.11	0.10
3	0.18	0.12
6	0.29	0.11
9	0.53	0.14
12	0.56	0.14
15	0.48	0.10
18	0.55	0.13
22	0.56	0.11
26	0.59	0.10
30	0.69	0.17

\* From Sommer et. al. (1982). Reprinted with permission from Archives of Environmental Contamination and Toxicology, Vol 11, p 592, Copyright 1982--Springer-Verlag. Values for fillet PCB concentration are the means of two to four fish, expressed as parts per million wet weight.

\*\* PCB concentrations were 0.2 and 1.8 ppm, respectively, in the control and experimental diets.



Table 38  
Accumulation of PCB in Goldfish Via Exposure to PCB-Saturated Water  
or Diet Containing 10 ppm PCB\*

Route of Exposure	Day	Wet Weight g	Extractable lipid, %	PCB Concentration, ppm Wet Weight				
				2,5- di	2,2',5'- tri	2,4',5'- tri	2,2',5,5'- tetra	2,3',4',5'- tetra
Water	0	5.5	4.1	1	--	--	--	--
	1	4.5	7.4	120	60	60	40	10
	2	4.3	6.4	290	160	110	50	10
	4	4.2	5.6	460	270	190	130	30
	8	5.4	5.1	1120	770	490	320	70
	13	3.2	9.2	2510	1820	1140	720	179
Mean Aqueous Concentration: $\mu\text{g}/\ell \pm 20\%$				190	110	75	55	22
Diet	10	2.8	1.8	0.321	0.536	0.714	0.643	0.821
	20	3.5	2.7	0.514	0.857	1.2	1.29	0.943
	30	7.3	1.9	0.274	0.274	0.411	0.521	0.548
	40	6.2	2.4	0.500	0.677	1.52	1.55	1.42
	50	3.2	3.3	0.313	0.469	0.563	0.719	0.625
	60	4.2	3.5	0.857	0.929	1.52	1.86	1.67
	70	3.3	3.4	0.879	0.485	1.45	1.15	1.24
	80	2.2	4.4	1.50	1.73	2.50	2.86	2.64
	90	5.3	2.4	0.434	0.472	1.51	1.70	1.79
	100	3.4	3.4	0.706	0.529	2.59	2.29	2.65
	110	4.9	3.0	0.469	0.367	1.43	1.39	1.63
	120	3.6	2.4	0.278	0.361	1.11	1.39	1.67
	130	6.1	2.8	0.393	0.459	1.41	1.57	1.72
	140	4.9	3.3	0.367	0.510	1.29	1.39	2.00
	150	3.8	2.5	0.553	0.658	1.97	2.13	2.79

\* From Bruggeman et al. (1981). Reprinted with permission from Chemosphere, Vol 10, pp 817, 822, W. A. Bruggeman, L. B. J. M. Martron, D. Kooiman, and O. Hutzinger, Accumulation and Elimination Kinetics of Di-, Tri-, and Tetra-Chlorobiphenyls by Goldfish after Dietary and Aqueous Exposure, Copyright 1981, Pergamon Press, Ltd.

Table 39  
Major Chlorocarbons in Fish, Lake Michigan--1971\*

Species	No. Fish Analyzed	Mean Fish Weight, g	Mean Lipid, %	PCB	DDE	ΣDDT
Alewife	85	100	6.5[3.9]	4.6[2.1]	1.7[0.8]	2.2[1.1]
Bloater	287	249	20.0[5.9]	6.0[2.2]	2.5[1.1]	3.8[2.8]
Brown trout	17	3,650	15.5[4.1]	7.3[2.8]	2.7[1.0]	4.2[1.6]
Carp	42	2,160	10.0[7.0]	4.2[3.6]	0.7[0.9]	0.9[1.2]
Chinook salmon	21	3,100	5.0[3.9]	11.4[4.0]	5.2[1.5]	6.8[2.5]
Coho salmon	56	2,720	6.5[2.1]	11.5[5.7]	4.8[2.3]	6.3[2.8]
Lake trout	134	1,620	16.6[4.3]	15.5[3.3]	5.0[2.8]	7.1[3.7]
Yellow perch	44	148	6.1[1.7]	5.8[3.5]	1.0[0.6]	1.6[1.1]
Rainbow trout	11	4,190	18.4[3.3]	9.3[4.1]	3.4[1.3]	4.2[1.8]
Redhorse sucker	16	902	8.6[1.2]	3.0[0.7]	1.6[0.5]	2.6[0.7]
Smelt	38	51	5.8[1.8]	2.7[1.3]	0.8[0.4]	1.2[0.6]
White sucker	51	1,130	5.9[2.8]	3.9[3.6]	1.0[0.5]	1.6[1.2]
Whitefish	43	1,170	17.6[4.4]	3.0[1.9]	0.8[0.3]	1.4[0.6]

\* After Veith (1975). Expressions in brackets represent standard deviations. Residues are expressed as parts per million wet weight.

Table 40

## Organochlorine and Mercury Residues in the Sediments and Biota of Lakes in Finland\*

Residue	Sediments	Plankton	Roach		Pike	
			Whole Tissue	Fat	Whole Tissue	Fat
			<u>Lake Uurainen</u>			
Hg (Total)	23 ± 27	--	176 ± 96	--	268 ± 606	--
(% Methyl)	--	--	94.8 ± 11.6	--	90.9 ± 11.48	--
DDE	--	119 ± 106	12.47 ± 6.06	1397 ± 872	7.02 ± 2.1	1100 ± 300
PCB	--	2054 ± 2283	58.1 ± 15.18	6504 ± 2295	39.7 ± 7.28	6500 ± 2300
HCB	--	6 ± 14	0.38 ± 1.2	33 ± 103	3.35 ± 2.12	500 ± 300
			<u>Lake Vätia</u>			
Hg (Total)	746 ± 99	--	260 ± 72	--	329 ± 57	--
(% Methyl)	--	--	90.3 ± 10.9	--	92.4 ± 6.77	--
DDE	1 ± 2	6 ± 9	5.05 ± 1.19	307 ± 137	3.81 ± 1.16	700 ± 200
PCB	361 ± 27	772 ± 834	154.7 ± 22.8	9101 ± 257	93.5 ± 47.1	15700 ± 5500
HCB	5 ± 4	1.5 ± 4	2.41 ± 1.03	127 ± 46	2.53 ± 2.12	500 ± 200
			<u>Päijänne</u>			
Hg (Total)	228 ± 88	--	238 ± 107	--	660 ± 95	--
(% Methyl)	--	--	85.1 ± 8.96	--	87.2 ± 8.4	--
DDE	14 ± 10	68 ± 97	5.15 ± 1.47	558 ± 170	11.5 ± 18.1	1900 ± 2500
PCB	209 ± 127	1490 ± 1201	85.48 ± 26.0	9269 ± 3050	109.3 ± 37.3	17200 ± 5700
HCB	0.4 ± 1	3.3 ± 8	0 ± 0	0 ± 0	4.5 ± 2.48	700 ± 400

\* After Paasivirta et al. (1983). Values are mean concentrations (parts per billion) ± standard deviations expressed on a dry weight basis in sediments and plankton and wet weight basis in fish.

Table 41  
Concentration Factors for PCB in Turbot Tissues Following  
Uptake of PCB from Different Sources\*

<u>External PCB Level on</u> <u>Day 1</u>	<u>Muscle (Liver) PCB</u> <u>Concentration, ppm</u> <u>on Day 14/15</u>	<u>Concentration</u> <u>Factor</u>
0.58 $\mu\text{g l}^{-1}$ seawater	2 (25)	$6 (50) \times 10^3$
100 ppm sediment	43 (469)	0.4 (4.7)
60 ppm sediment	59 (180)	1 (3)
1 ppm sediment	2 (22)	2 (22)
20 ppm food	4 (34)	0.2 (1.7)

\* From Courtney and Langston (1980). Reprinted with permission from Helgoländer Meeresuntersuchen, Vol 33, p 335, Copyright 1980--Biologische Anstalt Helgoland.

Table 42  
Uptake of Chlorinated Paraffins and PCB from Food by  
Juvenile Atlantic Salmon\*

Days of feeding		33		109		181	
Diet		Residue**	Lipid %	Residue**	Lipid %	Residue**	Lipid %
Control		0.30	1.03	nd†	0.65	nd†	0.47
Cereclor 42	10 µg/g	0.11	1.30	nd	0.69	nd	0.49
	100 µg/g	0.51	1.22	nd	0.49	nd	0.34
Chorez 700	10 µg/g	0.29	1.13	nd	0.40	nd	0.29
	100 µg/g	0.49	1.30	nd	0.56	nd	0.92
Aroclor 1254	10 µg/g	3.86††	5.09	3.80**	3.10	3.88**	2.07
	100 µg/g	13.9††	5.30	24.0**	2.73	30.0**	2.69

\* From Zitko (1974). Reprinted with permission from the Bulletin of Environmental Contamination and Toxicology, Vol 12, p 410, Copyright 1974--Springer-Verlag.

\*\* Expressed as chlorine, micrograms per gram wet weight, unless stated otherwise.

† Not detectable, <0.05 µg/g .

†† Expressed as Aroclor 1254.

Table 43

DDT, DDE, and PCB Residues (ppb) in the Biota of the Gulf of Mexico  
and Caribbean Sea During May and October 1971\*

Species and/or Common Name	May				October			
	DDT	DDE	Total DDT	PCB	DDT	DDE	Total DDT	PCB
<b>Invertebrates</b>								
Tunicate (colonial)	188	8.7	197	139				Whole
Sea pansy	128	161	289	850				Whole
Echinoderms (brittle stars)				8				Whole
Squid	65	4.6	70		4.6	4.3	9	Whole
Crustacean	86	16	102	151				Whole
( <i>Aristacus antillensis</i> )								
Shrimp (unidentified Penaeid)	26-154	7-18	33-165					Whole
( <i>Parapenaeus longirostris</i> )	304	34	338	167	1.0-5.2	2.1-2.7	3-8	Whole
Rock shrimp								Whole
Crabs ( <i>Callinectes</i> sp.)	1.8	7.4	9	17				Whole
Lobster ( <i>Nephropsis aculeate</i> )	151	6.0	157	22				Whole
<b>Smaller Fishes</b>								
Flounders ( <i>Syacium ounteri</i> )	3.6-84	3.3-10	7-94	32-34				Whole
( <i>S. papillosum</i> )	21	14	35	36-59				Whole
Flying fish ( <i>Exocoetidae</i> )	7.2	4.5	12	20	5.1-46	4.6-19	10-65	Whole
Squirrel fish	86	5.6	92	150				Whole
( <i>Holocentrus</i> sp.)								
Croakers		11	11	50				Whole
( <i>Micropogon undulatus</i> )								
Bat fish	12	65	77	527				Whole
<i>Paraques acuminatus</i>	20	6.6	27	27				Whole
<i>Peristedion oracile</i>	16	16	32	54				Whole
<i>Serranus atrobranchus</i>	33	8	41	68				Whole
<i>Saurida brasiliensis</i>	111	7	118	64				Whole
<i>Halicutichthys aculeatus</i>	78	4.8	83	64				Whole
<i>Trichossetta vontralis</i>	8.5	5.2	14					Whole
<i>Benthodesmus atlanticus</i>	91	4.3	95	36	4.5	10	15	Whole
<i>Synodus intermedius</i>								Whole
Unidentified species	33-141	7.3-18	40-159	53-56				Whole

(Continued)

\* After Giam et al. (1972). Values are reported as parts per million wet weight.

Table 43 (Concluded)

Species and/or Common Name	May			October		
	DDT	DDE	Total DDT	DDT	DDE	Total DDT
<b>Large Fishes</b>						
<b>Shark</b>						
( <i>Carcharinus falciformis</i> )	200	499	699			
( <i>C. springeri</i> )			1300			
<b>White tip shark</b>	15	15	30			
( <i>Pterolamiops longimanus</i> )	406	1100	1506			
<b>Tuna (<i>Euthynnus alletteratus</i>)</b>	44	31	75			
	45	39	84			
<b>King mackerel</b>	17	7.4	24			
( <i>Scomberomous cavalla</i> )	72	51	123			
<b>Parrot fish</b>	36		36			
( <i>Holichoeres radiatus</i> )						
<b>Red snapper (<i>Lutjanus aya</i>)</b>	3.9	9.5	13			
<b>Trigger fishes</b>	83	52	135			
( <i>Canthidermus sufflamen</i> )						
( <i>Balistes vetula</i> )						
<b>Yellowtailed snapper</b>	10	19	29			
( <i>Ocyurus chrysurus</i> )						
<b>Jack (<i>Thunnus atlanticus</i>)</b>						
<b>Barracuda</b>						
( <i>Sohygraana barracuda</i> )						
<b>Haemulon plumieri</b>						

Table 44  
Comparative Chlorinated Hydrocarbon Levels (ng g<sup>-1</sup> wet weight)  
in the Lower Medway Estuary, Kent\*

Organism		Location	DDT	Dieldrin	PCB
Mussel	<i>Mytilus edulis</i>	Medway	0-226	0-63	44-268
Crab	<i>Carcinus maenas</i>	Medway	0-50	0-23	0-142
Shrimp	<i>Crangon vulgaris</i>	Medway	2-82	1-21	0-275
Whiting	<i>Merlangius merlangus</i>	Medway muscle	5-17	0.6-6	16-96
		liver	630-4240	175-1096	2500-10500

\* After Wharfe and van den Broek (1978). Reprinted with permission from the Marine Pollution Bulletin, Vol 9, p 79, J. R. Wharfe and W. L. F. van den Broek, Baseline Concentrations of Polychlorinated Biphenyls and DDT in Lake Michigan Fish, 1971, Copyright 1975--Pergamon Press, Ltd.



Table 45  
Organochlorine Concentrations in Fish and Invertebrates from the  
Mediterranean Coast of Turkey\*

Species Analyzed	No. of Individuals Analyzed	% EOM**	Organochlorine Residue (ng g <sup>-1</sup> Fresh Weight)			
			t-DDE†	t-DDT	t-PCB††	
<i>Mugil auratus</i> (golden grey mullet)	20	1.8	min	5	8	T†
			max.	173	324	10
			mean	48	89	--
<i>Mullus barbatus</i> (striped mullet)	26	6.0	min.	2	9	T
			max.	122	257	2
			mean	62	130	--
<i>Mullus surmuletus</i> (red mullet)	6	2.6	min.	7	20	--
			max.	35	49	T
			mean	21	34	--
<i>Upeneus molleecensis</i> (gold band goat fish)	30	4.3	min.	31	49	T
			max.	69	94	T
			mean	47	74	--
<i>Parapanaeus kerathurus</i> (shrimp)	25	0.4	min.	3	4	T
			max.	61	65	T
			mean	28	34	T
<i>Patella caerulea</i> (limpet)	42	0.3	min.	1	2	2
			max.	4	7	39
			mean	2	5	15

\* From Basturk et al. (1980). Reprinted with permission from the Marine Pollution Bulletin, Vol 11, p 193, O. Basturk, M. Dogan, I. Salihoglu, and T. I. Balkas, DDT, DDE, and PCB Residues in Fish, Crustaceans, and Sediments from the Eastern Mediterranean Coast of Turkey, Copyright 1980--Pergamon Press, Ltd.

\*\* Percent extractable organic material, based on fresh weight of the living organism.

† t-DDE is the sum of op- and pp-DDE.

†† The PCB calculations were based on Aroclor 1254.

‡ Trace indicates less than 2 ng g<sup>-1</sup>.

Table 46  
PCB Concentrations in Cod and Some Important Cod Prey Organisms  
from Kiel Bay in 1977\*

Species	n	Mean Weight g	Lipid %	PCB Concentrations	
				ppb Wet Weight	ppm Lipid
<i>Diastylis rathkei</i>	696	0.013	0.81	169	20.65
<i>Pectinaria koreni</i> (scallop)	50	0.038	1.08	130	12.00
<i>Cyprina islandica</i> (clam) (soft body)	10	23.37	0.76	79	10.37
<i>Nephtys</i> spp. (polychaete)	80	0.083	0.82	80	9.83
<i>Crangon crangon</i> (shrimp)	40	0.292	0.51	32	6.34
	188	0.308	1.53	83	9.13
<i>Pomatoschistus minutus</i>	35	0.813	1.88	114	6.09
	178	0.751	3.03	109	3.58
<i>Sprattus sprattus</i> (sprat)	13	ca. 20.0	11.60	230	1.99
<i>Clupea harengus</i> (herring)	12	ca. 20.0	6.30	242	3.84
<i>Gadus morhua</i> (cod)					
liver	17	498.5	51.20	2874	5.74
fillet	18	482.0	0.73	23	3.17

\* From Schneider (1982). Reprinted with permission from Meeresforschung, Vol 29, p 72, Copyright 1982--Verlag Paul Parey.

Table 47  
DDT and PCB Concentrations (ppm) in Three California  
Marine Food Webs\*

Species	Trophic Level	Cs:K Ratio $\times 10^6$	Total PCB	Total DDT	PCB in Lipid	DDT in Lipid
<u>Coastal Pelagic Food Web</u>						
White shark	5.02	31.7	0.041	0.598	5.06	73.8
Mako shark	4.40	19.7	0.035	0.143	2.55	10.4
Sea lion	4.02	10.7	1.22	24.8	84.7	1722
Blue shark	4.00	13.7	0.016	0.107	1.86	12.4
Swordfish	3.97	12.3	0.020	0.105	0.76	2.63
Thresher shark	3.82	25.5	0.016	0.094	1.48	8.7
Bonita	3.80	8.79	0.029	0.184	2.79	17.7
Barracuda	3.74	4.20	0.115	0.800	2.17	11.2
Pacific mackerel	3.54	6.86	0.026	0.129	1.38	6.86
Market squid	3.62	3.32	0.012	0.014	<0.46	0.54
Pacific hake	3.09	8.58	0.012	0.036	1.56	4.68
Jack mackerel	3.04	5.73	0.014	0.119	1.12	9.52
Sardine	3.01	4.02	0.105	0.484	3.02	13.9
Basking shark	3.00	16.1	0.004	<0.005	--	--
Blue whale	3.00	11.0	0.003	0.050	0.20	3.33
Anchovy	2.82	1.87	0.008	0.047	0.500	2.94
Zooplankton	2.0	3.2	<0.003	0.011	0.278	1.08
<u>Palos Verdes Epibenthic Food Web</u>						
Scorpionfish	4.53	6.22	0.044	0.268	0.56	34.4
Spiny dogfish	4.16	31.3	5.12	65.5	41.9	537
Dover sole	3.52	3.62	0.279	7.17	16.1	414
White croaker	3.36	2.85	0.383	7.63	22.9	454
Ridgeback prawn	3.33	3.04	0.061	0.327	5.81	22.7
Mysids and decapods	2.78	13.8	0.032	0.323	2.22	22.4
Sediments	--	840	1.64	27.8	273	4633
<u>Eastern Tropical Pacific Food Web (Open Ocean)</u>						
Yellowfin tuna (50 kg)	4.82	13.3	<0.002	0.011	<0.217	1.20
Silky shark	4.81	22.8	<0.001	0.007	<0.115	0.804
Skipjack tuna	4.44	8.59	0.003	<0.001	0.462	<0.154
Yellowfin tuna (4 kg)	4.29	12.7	<0.002	<0.002	<0.308	<0.309
Frigate tuna	3.92	8.89	<0.001	0.005	<0.172	0.862
Squid	3.52	1.94	<0.001	<0.001	<0.169	<0.077
Flying fish	3.00	7.00	<0.002	<0.001	<0.235	<0.118
Zooplankton	2.00	3.30	0.002	<0.001	0.016	<0.083

\* After Schafer et al. (1982). Values are mean concentrations expressed as parts per million wet weight.

Table 48  
Concentrations of PCB in Cottids, Mussels, and Sediment from  
Southern Puget Sound\*

<u>Site No.</u>	<u>Concentration in Cottids ppb Wet Wt.</u>	<u>Concentration in Mussels ppb Wet Wt.</u>	<u>Concentration in Sediment ppb Dry Wt.</u>
1	65	85	1.8
2	470	95	70
3	840	210	330
4	180	31	1.0
5	200	72	6.3
6	500	38	17
7	100	16	1.5
8	130	50	2.6
9	56	27	1.1
10	66	30	2.1
11	29	14	0.7
12	29	24	1.5
13	62	11	3.1
14	21	16	0.7
15	29	10	5.7
16	63	11	1.8
17	160	14	10
18	190	40	7.3

\* After Mowrer et al. (1977). Reprinted with permission from the Bulletin of Environmental Contamination and Toxicology, Vol 18, p 592, Copyright 1977--Springer-Verlag.

Table 49  
Concentrations of Organochlorine Residues in Five Species of Littoral  
Animals in the Weser Estuary\*

Species and Age Group	No. of Animals	Average Length cm	Average Wet Weight** g	Dry Weight %	Lipid %	Residue Concentration, ng g <sup>-1</sup>						
						PCB	p,p'- DDD	Dieldrin	α-HCH	γ-HCH	p,p'- DDE	α-Endo- smt
<i>Cerastoderma edule</i> (L.), common edible cockle (Bivalvia) II	80	3.6	4.3	20.2	0.31	7	0.9	1.6	1.0	2.8	0†	
	80	3.6	4.1	20.2	0.31	11	0.9	2.2	1.1	2.7	0	0.2
	81	3.6	4.1	20.7	0.29	10	0.6	1.4	0.7	2.5	0	0.3
	78	3.7	4.1	19.0	0.33	9	0.8	1.8	0.7	1.8	0	0.3
	80	3.6	4.1	20.8	0.35	17	1.0	0.6	0.9	2.4	0	0.2
	$\bar{x}$	3.6	4.1	20.2	0.32	11	0.8	1.5	0.9	2.4	0	0.3
sd		±0.2	ND††	±0.8	±0.33	±4 (3400 ±1000)	±0.2 (260 ±40)	±0.6 (490 ±200)	±0.2 (280 ±60)	±0.4 (770 ±150)	±0 (0 ±0)	±0.1 (80 ±30)
<i>Mya vernaria</i> L., soft clam (Bivalvia) VI	10	8.9	35.1	23.3	1.7	42	6.9	7.2	3.7	3.5	0.9	0.8
	10	9.0	33.1	23.4	1.7	59	5.2	4.3	3.5	4.5	1.1	0.8
	10	9.1	37.8	22.9	1.5	42	3.7	4.3	3.0	2.0	0.9	0.6
	10	9.2	31.4	23.8	1.7	54	4.8	3.8	2.7	1.6	0.7	0.5
	10	9.0	34.9	22.8	1.6	23	4.4	3.0	2.7	2.4	0.8	0.6
	$\bar{x}$	9.0	34.4	23.2	1.6	44	5.0	4.5	3.1	2.8	0.9	0.7
sd		±0.6	ND	±0.4	±0.1	±14 (2700 ±800)	±1.2 (300 ±70)	±1.6 (270 ±100)	±0.5 (190 ±30)	±1.2 (170 ±70)	±0.2 (54 ±9)	±0.2 (40 ±10)

(Continued)

\* From Goerke et al. (1979). Reprinted from the Marine Pollution Bulletin, Vol 10, H. Goerke, G. Eder, K. Weller, and W. Ernst, Patterns of Organochlorine, Residues in Animals of Different Trophic Levels from the Weser Estuary, Copyright 1979--Pergamon Press, Ltd. Values are on a wet tissue basis or (in parentheses) on a lipid basis.

\*\* Soft tissues for bivalvia.

† Since PCB interference prohibits the direct evaluation of the p,p'-DDE peak, there is no general detection limit for p,p'-DDE.

†† ND = not determined.

Table 49 (Concluded)

Species and Age Group	No. of Animals	Average Length cm	Average Wet Weight g	Dry Weight %	Lipid %	Residue Concentration, ng g <sup>-1</sup>						
						PCB	P,p'- DDD	Dieldrin	$\alpha$ -HCH	$\gamma$ -HCH	P,p'- DDE	$\alpha$ -Endo- sulfan
<i>Arenicola</i>	20	ND	14.7	22.2	1.3	62	3.2	ND	1.4	2.2	1.0	0.3
<i>Marina</i> (L.),	20	ND	14.0	21.1	1.3	57	3.1	ND	1.6	1.6	0.8	0.5
Lugworm	20	ND	13.5	20.8	1.5	91	4.5	ND	1.5	1.1	1.2	0.5
(Polychaeta)	20	ND	14.7	22.3	1.4	57	3.9	ND	1.3	2.1	0.8	0.3
II	20	ND	13.7	20.6	1.5	51	4.0	ND	1.5	1.5	1.0	0.5
$\bar{x}$			14.1	21.4	1.4	64	3.7		1.5	1.7	1.0	0.4
sd			$\pm 2.6$	$\pm 0.8$	$\pm 0.1$	$\pm 16$	$\pm 0.6$		$\pm 0.2$	$\pm 0.5$	$\pm 0.2$	$\pm 0.2$
						(4500)	(270)		(100)	(120)	(69)	(30)
						$\pm 1000$ )	$\pm 30$ )		$\pm 20$ )	$\pm 40$ )	$\pm 10$ )	$\pm 8$ )
<i>Crangon</i>	100		2.6	20.8	1.2	59	0.8	0.6	2.8	2.7	0.9	0.4
<i>crangon</i> L.,	100	6.0	2.3	23.9	1.3	64	0.6	0.5	2.4	1.6	0.9	0.3
brown shrimp	100	to	2.4	23.8	1.3	56	0.8	0.4	2.0	1.4	0.9	0.3
(Crustacea)	100	6.5	2.2	23.8	1.3	62	0.6	0.5	2.2	1.4	0.9	0.3
I-II	100		2.3	24.3	1.4	58	0.6	0.5	2.3	1.9	0.9	0.3
$\bar{x}$			2.4	23.3	1.3	60	0.7	0.5	2.3	1.8	0.9	0.3
sd			ND	$\pm 1.5$	$\pm 0.1$	$\pm 4$	$\pm 0.2$	$\pm 0.1$	$\pm 0.3$	$\pm 0.6$	$\pm 0.0$	$\pm 0.1$
						(4600)	(50)	(40)	(180)	(140)	(70)	(20)
						$\pm 400$	$\pm 20$	$\pm 10$	$\pm 40$	$\pm 50$	$\pm 10$	$\pm 10$ )
<i>Solea solea</i> L.,	8	16.5	37.6	24.5	2.8	218	10.0	4.4	5.4	3.4	4.5	1.4
common sole	8	16.6	37.2	24.5	2.7	281	13.4	5.2	4.7	4.3	4.6	1.7
(Pisces)	8	16.7	39.0	23.7	2.2	173	7.5	2.8	2.9	2.0	2.8	0.9
I	8	16.8	41.9	24.5	2.8	204	7.7	3.6	4.0	3.3	3.5	0.9
	8	16.6	36.7	24.7	3.0	155	10.7	4.8	3.6	2.4	3.6	1.4
$\bar{x}$		16.6	38.5	24.4	2.7	206	9.9	4.2	4.1	3.1	3.8	1.3
sd		$\pm 1.0$	$\pm 7.8$	$\pm 0.4$	$\pm 0.3$	$\pm 49$	$\pm 2.5$	$\pm 1.0$	$\pm 1.0$	$\pm 0.9$	$\pm 0.8$	$\pm 0.4$
						(7700)	(370)	(150)	(150)	(110)	(140)	(47)
						$\pm 1900$	$\pm 90$	$\pm 30$	$\pm 40$	$\pm 40$	$\pm 30$	$\pm 12$

Table 50  
DDE, PCBs, and HCB in Sediments and Biological Samples from Ora, Norway\*

Species According to Feeding Habits	No. Analyzed	Fat, %	ppb Wet Weight		ppm Fat Weight	
			DDE	PCB	DDE	PCB
Bottom sediments	3		1	24	2	0.002
Filter feeders						
Barnacles	2		12	114		
Mussels	2	0.8	5	59	2	0.63
Detritus feeders						
Bristle worm	2	2.6	10	100		3.8
Shrimp	4	3.1	7	98		3.2
Amphipod/Nereis feeders						
Common gobi	5	3.1	13	217	18	0.43
Flounder	15	3.5 ±0.9	23 ±9	325 ±66	67 ±30	9.1 ±0.3
Scavenger						
Crab	9		18 ±12	187 ±64	26 ±26	

\* From Bjerk and Brevik (1980). Reprinted with permission from Archives of Environmental Contamination and Toxicology, Vol 7, p 745, Copyright 1980--Springer-Verlag. Values are means ± standard deviations.

Table 51  
Organochlorine Content (ppm) of Flounder and Cockles\*

Organs	% Fat	PCB	DDE	DDD	DDT	DDT
<i>Cerastoderma edule</i> (cockle)						
(n = 20)						
	2.57 ± 0.72	0.06 ± 0.03	0.0014 ± 0.0006	0.0011 ± 0.0002	0.0008 ± 0.0002	0.0033 ± 0.0007
<i>Platichthys flesus</i> (flounder)						
(n = 10)						
Liver	34.0 ± 12.3	5.71 ± 4.39	0.30 ± 0.20	0.30 ± 0.23	0.19 ± 0.18	0.77 ± 0.50
Gills	10.44 ± 4.89	1.05 ± 0.55	0.07 ± 0.05	0.07 ± 0.04	0.06 ± 0.04	0.20 ± 0.08
Muscle	4.94 ± 6.47	0.60 ± 0.45	0.03 ± 0.02	0.04 ± 0.03	0.04 ± 0.03	0.11 ± 0.06
Gonad	10.91 ± 11.66	0.57 ± 0.98	0.03 ± 0.04	0.02 ± 0.02	0.03 ± 0.03	0.07 ± 0.08
Gut	4.93 ± 2.38	0.50 ± 0.36	0.04 ± 0.03	0.02 ± 0.01	0.03 ± 0.02	0.08 ± 0.04

\* From Courtney and Langston (1980). Reprinted with permission from Helgoländer Meeresuntersuchen, Vol 33, p 337, Copyright 1980--Biologische Anstalt Helgoland. Values are means ± standard deviations, expressed as parts per million wet weight.



Table 52  
Concentrations of PCBs Detected in the Brisbane River Ecosystem\*

Sample Identification and Number Analyzed	Sample Type**	PCB, ppm	
		Mixture Containing 54% Cl	Mixture Containing 60% Cl
Water (8), ppb	--	ND-0.009†	ND-0.05
Sediments (8)	--	ND-0.054	ND-0.058
Crabs	w,wwb	ND-0.05	0.034-0.26
( <i>Sesarma erythrodactyla</i> ) (7)	w, lb	ND-2.0	ND-23
Mud Crab	m, ww	ND	0.09
( <i>Helograpsus haswellianus</i> ) (1)	m, lb	ND	38
Polychaetes	w, ww	ND-0.23	ND-0.29
(Unidentified) (5)	w, lb	ND-1.9	ND-13
Mussels	w, ww	ND-0.052	ND-0.25
( <i>Mytilus coruscus</i> ) (2)	w, lb	ND-3.8	ND-45
Periwinkles	w, ww	0.03	0.015
( <i>Austrocochlea obtusa</i> ) (1)	w, lb	8	4
Oyster Blennies	w, ww	0.10-0.13	ND-1.4
( <i>Petroscirtes anolius</i> ) (3)	w, lb	2.2-7.2	ND-60
Whiting	m, ww	0.07-0.71	0.22-0.70
( <i>Sillago ciliata</i> ) (2)	m, lb	6.9-30	20-30
Striped Butterfish	m, ww	0.026	0.041
( <i>Selenotoca multifasciata</i> ) (1)	m, lb	2.6	4.1
Bony Bream	m, ww	0.022-0.040	0.14-0.33
( <i>Nematolosa come</i> ) (2)	m, lb	2.5-5.5	20-34
Flathead	m, ww	0.014	0.033
( <i>Platycephalus fuscus</i> ) (1)	m, lb	17	41
Mullet	m, ww	ND-0.94	ND-2.9
( <i>Mugil cephalus</i> ) (15)	m, lb	ND-23	ND-230
Pelican	m, ww	ND-1.7	2.1-15.7
( <i>Pelecanus conspicillatus</i> ) (6)	m, lb	ND-36	45-350

\* From Shaw and Connell (1980). Reprinted with permission from Marine Pollution Bulletin, Vol 11, G. R. Shaw and D. W. Connell, Polychlorinated Biphenyls in the Brisbane River Estuary, Australia, Copyright 1980--Pergamon Press, Ltd.

\*\* w = whole organism analyzed; lb = lipid basis concentration; ww = wet weight basis concentration; m = muscle tissue; g = gill tissue.

† ND = not detected.

Table 53  
Concentration Factors (in Comparison to Water) for  
DDT Bioaccumulation\*

<u>Location</u>	<u>Year of Study</u>	<u>Component</u>		
		<u>Algae</u>	<u>Invertebrates</u>	<u>Fish</u>
Farm pond	1965	20,000	28,750	363,000
Farm pond	1966	5789	27,868	317,000
Pools	1967			
	0.05 ppm	7700	20,000	78,500
	1.0 ppm	1843	11,687	76,006
	10.0 ppm	770	5,941	23,125
Mean		7220	18,849	171,526
Standard Deviation		7683	9,982	156,225

\* From Hamelink, Waybrant, and Ball (1971). Reprinted with permission from the Transactions of the American Fisheries Society, Vol 100, p 212, Copyright 1971--American Fisheries Society.

Table 54  
Chlorinated Hydrocarbon Pesticides in the Biota at  
Tule Lake Wildlife Refuge\*

Component	Date Collected	DDE	DDD/DDT**	Chlordane	Endrin
Suspended material	4/20/66	--	0.75	3.0	1.5
	6/22/66	--	--	67.0	6.0
	7/22/66	1.7	10.0	6.0	1.3
	8/22/66	6.6	4.0	6.0	57.7
	9/13/66	--	4.0	8.0	13.0
	10/26/66	1.0	0.7/2.0	1.5	5.3
	11/16/66	--	--	8.5	--
	1/06/67	1.5	3.3/12.0	14.7	1.5
Vascular plants	6/22/66	1.0	1.0	5.0	--
	7/22/66	--	--	2.0	1.6
	8/22/66	1.0	2.0	2.0	12.2
	9/13/66	--	--	1.5	12.5
	9/29/66	0.8	1.2	--	4.8
	10/20/66	1.0	10.0	6.0	8.0
	11/16/66	0.6	0.7	2.6	1.8
Algae	4/20/66	0.5	0.75	2.0	2.0
	6/22/66	2.0	3.0	50.0	--
	7/22/66	--	--	--	--
	8/22/66	0.8	0.4	1.7	22.3
	9/13/66	1.3	1.3	13.5	10.8
Clams	8/10/65	4.0	4.0	3.0	34.0
	12/28/65	4.0	3.0	4.5	4.0
	7/22/66	4.8	4.8	12.0	2.0
Chubs	8/27/65	45.0	17.0	--	198.0
	4/20/66	26.0	12.0	24.0	10.0
	6/22/66	14.0	10.0	10.0	6.0
	7/22/66	6.2	9.6	8.0	4.0
	8/22/66	2.5	2.5	--	30.5

\* From Godsil and Johnson (1968). Values are reported as parts per billion wet weight. The indication -- means levels were below analytical sensitivity.

\*\* Single values represent a total response, i.e., where DDD and DDT could not be separated.

Table 55  
Residues of DDT (ppm) in Whole Fish of Cayuga Lake at Taughannock,  
New York, in October 1963\*

<u>Species</u>	<u>Total DDT</u>
Sunfish	0.2
Rock bass	0.3
White sucker	0.5
Alewife	0.7
Smallmouth bass	0.7
Yellow perch	1.1
Lamprey	5.3
Lake trout (immature)	3.6
Lake trout (mature)	6.2

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\* After Mack et al. (1964). Values are means expressed as parts per million wet weight.

Table 56  
Chlorinated Hydrocarbon Concentrations (ppm) in Tissues of Fish from Lakes in the  
Vicinity of Cold Lake, Alberta, in 1978\*

Species	N	PCB, Mean (Range)	HCB, Mean (Range)	α BHC, Mean (Range)	pp'DDT, Mean (Range)	op'DDT, Mean (Range)	DDE, Mean (Range)	DDD, Mean (Range)	Σ DDT, Mean (Range)
<u>Muscle</u>									
Cisco (Lake Herring)	13	0.002 (ND-0.006)	0.001 (T-0.002)	0.002 (T-0.005)	T (ND-T)	T (ND-0.002)	0.007 (T-0.035)	T (ND-0.002)	0.007+ (T-0.035)
Whitefish	12	0.002 (ND-0.157)	0.001 (T-0.021)	0.004 (T-0.042)	0.001 (ND-0.004)	T (ND-0.002)	0.011 (T-0.038)	0.002 (ND-0.016)	0.014+ (T-0.050)
White Sucker	8	0.001 (ND-0.004)	T (T-1.001)	0.001 (T-0.002)	T (ND-0.001)	ND	0.002 (ND-0.009)	T (ND-0.003)	0.002+ (T-0.009)+
Pike	11	0.002 (ND-0.008)	T (ND-T)	T (T-0.003)	T (ND-0.005)	T (ND-T)	0.002 (ND-0.013)	0.001 (ND-0.006)	0.004+ (T-0.016)
<u>Fat</u>									
Cisco (Lake Herring)	6	0.113 (ND-0.939)	0.055 (0.03-0.195)	0.098 (0.037-0.274)	0.018 (ND-0.037)	0.022 (ND-0.127)	0.458 (0.05-1.589)	0.111 (ND-0.402)	0.609 (0.054-2.154)
Whitefish	12	0.274 (ND-2.600)	0.047 (0.009-0.213)	0.071 (0.027-0.278)	0.047 (ND-0.458)	0.026 (ND-0.216)	0.591 (0.026-5.252)	0.082 (NI-0.350)	0.746 (0.03-6.230)
White Sucker	4	0.057 (0.001-0.162)	0.023 (T-0.048)	0.051 (T-0.104)	0.016 (0.001-0.035)	0.004 (ND-0.015)	0.117 (0.005-0.211)	0.069 (0.002-0.152)	0.206 (0.008-0.382)
Pike	9	0.55 (0.101-1.703)	0.034 (0.006-0.086)	0.084 (0.044-0.162)	0.135 (0.004-0.861)	0.017 (ND-0.110)	0.907 (0.044-4.520)	0.148 (0.011-0.711)	1.207 (0.07-4.607)

\* From Tsui and McCart (1981). Reprinted with permission from the International Journal of Environmental Analytical Chemistry, Vol 10, p 281, Copyright 1981--Gordon and Breach Science Publishers Limited. Values are expressed as parts per million wet weight of muscle or as parts per million in fats. ND = not detectable, T = trace [less than 0.001 ppm].

Table 57  
Average Percent Fat Content and Mean Whole-Body Insecticide Levels in Fish  
Collected in July and October 1978 in the Des Moines River, Iowa\*

<u>Fish Species</u>	<u>No. Fish</u>	<u>Fat %</u>	<u>Concentration, ppm</u>		
			<u>Dieldrin</u>	<u>DDT</u>	<u>Heptachlor Epoxide</u>
Gizzard shad	104	17.0	81 (0.48)	55 (0.32)	14 (0.08)
River carpsucker	72	5.4	45 (0.83)	39 (0.72)	3 (0.06)
Carp	91	4.5	35 (0.78)	45 (1.00)	6 (0.13)
Channel catfish	3	8.5	101 (1.19)	101 (1.19)	10 (0.12)
White crappie	79	2.7	56 (2.07)	46 (1.70)	8 (0.30)
Walleye	26	2.0	26 (1.30)	106 (5.30)	2 (0.10)
Largemouth bass	52	4.0	61 (1.52)	75 (1.87)	8 (0.20)

\* From Bulkley, Leung, and Richard (1981). Values are expressed on the basis of parts per million wet weight and (in parentheses) parts per million in fat.

Table 58  
DDE Residues (ppm) in Three Kenya Lake Drainage Systems\*

Organism	Drainage System		
	Naivasha	Nakuru	Baringo
Vegetation			
Algae	0.007 ± 0.020	Trace	ND-Trace
Higher aquatics	0.030 ± 0.067	--	ND-Trace
Ferns	0.107 ± 0.035	--	--
Insects	0.034 ± 0.076	Trace	0.090 ± 0.127
Crustacea	--	0.04 ± 0.20	--
Other invertebrates	Trace	--	--
Fish			
<i>Tilapia</i>	0.007 ± 0.011	0.074 ± 0.051	0.043 ± 0.015
Black bass	0.016 ± 0.019		
<i>Clarias mosambicus</i>	--	--	0.095 ± 0.033
<i>Barbus gregorii</i>	--	--	0.143 ± 0.080
<i>Labeo cylindricus</i>	--	--	2.13

\* After Lincer et al. (1981). Values are expressed as parts per million dry weight.

Table 59  
DDT and PCB Concentrations (ppm) in Three California Saltwater Food Webs\*

Species	Common Name	Estimated Trophic Level	Cs:K Ratio x 10 <sup>6</sup>	% Lipids	Total DDT	Total PCB
	<u>Salton Sea Food Web (Inland)</u>					
<i>Cynoscion xanthulus</i>	Orangemouth corvina	IV-V	32.0	2.0	0.20	0.014
<i>Bairdiella icistia</i>	Gulf croaker	III-IV	20.9	1.8	0.064	0.002
<i>Anisotermus elavidsoni</i>	Sargo	III-IV	18.8	8.0	0.19	0.008
<i>Dorosoma petenense</i>	Thre.dfin shad	III	10.0	9.9	0.48	0.028
<i>Poecilia latipinna</i>	Sailfin molly	II-III	14.3	5.5	0.040	0.000
	<u>Newport Bay Food Web</u>					
<i>Morone saxatilis</i>	Striped bass	IV-V	4.94	0.91	0.75	0.29
<i>Paralabrax maculatofasciatus</i>	Spotted sand bass	IV-V	5.51	1.07	0.48	0.19
<i>Umbrina roncadore</i>	Yellowfin croaker	III-IV	5.53	1.2	0.2	0.042
<i>Atherinops affinis</i>	Topsmelt	III	3.69	0.67	0.15	0.039
<i>Mulgil cephalus</i>	Striped mullet					
	(large)	II	4.47	8.6	4.4	0.47
	(small)	II	3.59	4.0	1.00	0.12
	<u>Palos Verdes Food Web</u>					
<b>Fishes</b>						
<i>Sebastes paucispinis</i>	Bocaccio	IV-V	16.6	1.47	0.61	0.072
<i>Scorpaena guttata</i>	Scorpionfish	IV-V	13.6	0.69	3.5	0.39
<i>Citharichthys sordidus</i>	Sandlab	III-IV	12.1	0.88	6.1	0.38
<b>Crustaceans</b>						
<i>Cancer anthonyi</i>	Yellow crab	III-IV	6.5	0.52	1.5	0.19
<i>Sicyonia ingentis</i>	Ridgeback prawn	III-IV	11.2	1.27	0.15	0.058
<b>Molluscs</b>						
<i>Hinnites multirugosus</i>	Scallop	II-III	5.4	0.76	0.16	0.012
<i>Haliotis cracherodii</i>	Black abalone	II	7.6	0.94	0.001	0.006

\* After Young and Mearns (1979). Metal concentrations based on parts per million in wet muscle tissues.



Table 60  
Concentrations of Organochlorine Compounds in Marine Samples  
Taken Off the Northumberland Coast, 1965-1966\*

Species	Trophic Level	Type of Tissue and Number of Samples	Concentration of Organochlorine Compounds, ppm	
			HEOD	pp'-DDE
Serrated wrack ( <i>Fucus serratus</i> )	1	Pooled sample	0.001	0.002
Oar weed ( <i>Laminaria digitata</i> )	1	Pooled sample	0.001	0.003
Microzooplankton	2	Pooled sample	0.020	0.030
Sea urchin ( <i>Echinus esculentus</i> )	2	Pooled sample	0.027	0.050
Mussel ( <i>Mytilus edulis</i> )	2	Pooled sample of flesh	0.023	0.024
Cockle ( <i>Cardium edule</i> )	2	Pooled sample of flesh	0.018	0.012
Limpet ( <i>Patella vulgata</i> )	2	Pooled sample of flesh	0.009	0.003
Microzooplankton (Crustacea)	3	Pooled sample	0.16	0.16
Lobster ( <i>Homarus vulgaris</i> )	3	Pooled flesh from two specimens	0.024	0.024
Shore crab ( <i>Carcinus maenas</i> )	3	Pooled flesh from two specimens	0.025	0.037
Edible crab ( <i>Cancer pagurus</i> )	3	Pooled flesh from two specimens	0.015	0.061
Plaice ( <i>Pleuronectes</i> sp.)	3	Whole fish	0.038	0.023
Herring ( <i>Clupea harengus</i> )	3	Pooled sample of whole fish	0.057	0.080
Sand eels ( <i>Ammodytes lanceolatus</i> )	3	Twelve samples of whole fish	0.016 (0.004-0.021)	0.026 (0.007-0.031)
Cod ( <i>Gadus morhua</i> )	4	Six samples, three fish per sample	0.009 (0.002-0.018)	0.012 (0.004-0.024)
Whiting ( <i>G. merlangus</i> )	4	Pooled sample of whole fish	0.040	0.021

\* After Robinson et al. (1967). Reprinted by permission from *Nature*, Vol 214, p 1308, Copyright 1967--MacMillan Journal Limited. Values in parentheses are the 95% confidence limits of the means.

Table 61  
DDT Residues (DDT + DDE + DDD) in Samples from Carmans River Estuary  
and Vicinity, Long Island, N. Y.\*

Sample	DDT Residues ppm	Percent of Residues		
		DDT	DDE	DDD
Water	0.0005			
Plankton (mostly zooplankton)	0.040	25	75	Trace
<i>Cladophora gracilis</i>	0.083	56	28	16
Shrimp	0.16	16	58	26
<i>Opsanus tau</i> , oyster toadfish (immature)	0.17	None	100	Trace
<i>Menidia menidia</i> , Atlantic silverside	0.23	17	48	35
<i>Nassarius obsoletus</i> , mud snail	0.26	13	39	43
<i>Gasterosteus aculeatus</i> , threespine stickleback	0.26	24	51	25
<i>Anguilla rostrata</i> American eel (immature)	0.28	29	43	28
Flying insects, mostly Diptera	0.30	16	44	40
<i>Spartina patens</i> , shoots	0.33	58	26	16
<i>Mercenaria mercenaria</i> , hard clam	0.42	71	17	12
<i>Cyprinodon variegatus</i> , sheepshead minnow	0.94	12	20	68
<i>Fundulus heteroclitus</i> , mummichog	1.24	58	18	24
<i>Paralichthys dentatus</i> , summer flounder	1.28	28	44	28
<i>Esox niger</i> , chain pickerel	1.33	34	26	40
<i>Strongylura marina</i> , Atlantic needlefish	2.07	21	28	51
<i>Spartina patens</i> , roots	2.80	31	57	12

\* From Woodwell, Wurster, and Isaacson (1967). Reprinted by permission from *Science*, Vol 156, p 822, Copyright 1967 by the AAAS. Values for DDT expressed as parts per million wet weight of whole organisms.

Table 62  
Organochlorine Residues in Swedish Marine Organisms, 1965-68\*

Organism	No. in Sample	ppm in Fat		ppm in Fresh Tissue		Percent Fat
		DDT	PCB	DDT	PCB	
Swedish West Coast						
Mussel	17	1(0.4-5)	2(0.5-7.0)	0.02(0.005-0.04)	0.084(0.011-0.33)	1.3(0.66-2.6)
Plaice	3	1(0.9-2)	5(0.4-14)	0.006(0.003-0.009)	0.021(0.002-0.056)	0.5(0.4-0.5)
Cod	4	1(0.6-2)	7.3(1.8-16)	0.005(0.001-0.006)	0.019(0.006-0.030)	0.30(0.19-0.34)
Picked dogfish	7	1.5(0.29-3.9)	1.5(0.81-2.4)	0.15(0.028-0.33)	0.15(0.054-0.30)	9.6(6.7-14)
Baltic Sea Proper, Including the Sound						
Mussel	40	6(0.9-10)	4.3(1.9-8.6)	0.03(0.009-0.07)	0.03(0.008-0.057)	0.92(0.46-1.6)
Herring	18	17(4.1-37)	6.8(0.5-23)	0.68(0.093-2.3)	0.27(0.009-1.0)	4.4(0.7-12)
Plaice	6	2.7(1.4-7.8)	2.7(1.7-4.8)	0.018(0.006-0.036)	0.017(0.010-0.032)	0.65(0.58-0.71)
Cod	5	19(12-31)	11(3.2-20)	0.063(0.027-0.11)	0.033(0.012-0.057)	0.32(0.23-0.44)
Salmon	11	31(20-53)	2.9(1.1-8.2)	3.4(0.26-7.1)	0.30(0.014-0.54)	11(1.2-20)
The Archipelago of Stockholm						
Mussel	15	3(1-4.7)	5.2(3.4-7.0)	0.004(0.01-0.061)	0.037(0.032-0.044)	1.1(0.94-1.3)
Herring	4	7.7(4.3-11)	5.1(3.3-8.5)	0.23(0.094-0.30)	0.17(0.073-0.23)	2.6(2.2-2.8)

\* After Jensen et al. (1969).

Table 63  
Transfer of Dieldrin From Clams to Crabs\*

<u>Test</u>	<u>Treatment</u>	<u>No. Days of Feeding</u>	<u>Dieldrin in Clam ppb</u>	<u>Dieldrin in Crab, ppm <math>\bar{X} \pm sd</math></u>	<u>Range</u>
1	Control	10	1	$7.5 \pm 1.9$	(5.9 - 8.4)
	Experimental	10	193	$25.3 \pm 4.6$	(19.2 - 31.9)
2**	Control	10	1	$4.7 \pm 1.5$	(2.8 - 6.9)
	Experimental	10	181	$96.6 \pm 78.1$	(18.4 - 203.9)
3	Control	5	1	$3.4 \pm 1.3$	(2.4 - 5.9)
	Experimental	5	193	$32.8 \pm 5.5$	(29.3 - 41.5)

\* After Petrocelli, Anderson, and Hanks (1975).

\*\* The weights of the crabs in this test were about one fourth of those in the other two tests.

Table 64  
Organochlorine Insecticide Residues in Fat from Fishes in  
the Transvaal, South Africa\*

Species	Number	Locality**	Residues (mg/kg Fat)					Total
			Total BHC	p,p'- DDE	p,p'- DDT	Dieldrin	Total Endosulfan	
Barbel	3	A	0.38	28.80	1.52	1.33	1.24	33.27
	1	B	0.28	7.50	0.83	0.38	1.70	10.68
	1	C	0.55	7.71	ND†	0.90	0.33	9.49
		$\bar{X}$	0.40	14.67	0.78	0.87	1.09	17.82
Kurper	1	A	0.37	18.63	5.65	1.44	ND	26.09
	1	B	0.17	5.34	0.11	0.41	0.50	6.53
	1	C	0.16	4.85	0.68	0.50	ND	6.19
		$\bar{X}$	0.23	9.61	2.15	0.78	0.17	12.94
Yellowfish	1	A	0.26	9.18	ND	ND	5.88	15.32
Mudfish	1	D	0.19	3.79	0.43	0.51	ND	4.92

\* After Pick, de Beer, and van Dyk (1981). Reprinted with permission from Chemosphere, Vol 10, Pick, F. E., P. R. de Beer, and L. P. van Dyk, Organochlorine Insecticide Residues in Birds and Fish from the Transvaal, South Africa, Copyright 1981--Pergamon Press, Ltd.

\*\* A = Olifants River near Marble Hall; B = Olifants River Dam near Phalaborwa; C = Letaba River in Hans Merensky Nature Reserve; D = Crocodile River near Kaap Muiden.

† ND = not detectable.

Table 65  
Kepone Transfer in a Plankton-Mysid-Fish Food Chain\*

Variables	Control Food Chain	Low Exposure Food Chain	High Exposure Food Chain
1. Kepone (single dose) in brine shrimp media (mg/l)	Control	0.005	0.1
2. Kepone residues in brine shrimp after 48 hrs of exposure (mg/kg)	Control (ND)**	0.049 0.043 0.058 $\bar{X} = 0.050$	1.3 2.4 3.3 $\bar{X} = 2.33$
3. Bioconcentration factor from water [(2)/(1)]	--	10	23.3
4. Kepone residues in mysids after 72 hours of feeding (mg/kg)	Control (ND)**	$\bar{X} = 0.023$ (estimated)	0.89 1.0 1.8 $\bar{X} = 1.23$
5. Bioaccumulation factor from brine shrimp to mysids [(4)/(2)]	--	0.5 (estimated)	0.53
6. Kepone residues in spot after 30 days of feeding (mg/kg)	Control (ND)**	0.015 0.024 $\bar{X} = 0.0195$	1.0 1.1 $\bar{X} = 1.05$
7. Bioaccumulation factor from mysids to spot [(6)/(4)]	--	>0.85 (estimated)	>0.85
8. Food chain factor [(6)/(1)]	--	>3.9	>10.5

\* From Bahner et al. (1977). Reprinted with permission from Chesapeake Science, Vol 18, p 307, Copyright 1977--Estuarine Research Foundation. Brine shrimp (*A. salina*) were hatched during 48 hr in kepone-enriched sea-water and were fed to mysids (*M. bahia*) for 72 hr. Mysids were then fed to spot (*L. xanthurus*) for 30 days in flow-through feeding experiment.

\*\* ND = non-detectable (<0.02 mg/kg).

Table 66

## Chlorinated Phenols\* (ppb) in the Biota of Lakes\*\* in Finland†

Population or Species	Station	N	Trichl. ph.			Tetrachl. ph.			Pentachl. ph.			Trichl. gua.			Tetrachl. gua.			Tetrachl. cat.		
			$\bar{X}$	sd	ph.	$\bar{X}$	sd	ph.	$\bar{X}$	sd	ph.	$\bar{X}$	sd	ph.	$\bar{X}$	sd	ph.	$\bar{X}$	sd	ph.
Pike	K	8	0.79	1.6		20.2	40.0		6.49	10.5		2.08	5.87		1.74	3.74		3.99		5.67
	P	6	17.3	18.1		11.1	14.9		5.72	10.1		16.4	18.5		40.4	68.7		10.2		13.9
	V	10	13.6	19.1		19.0	12.4		8.04	5.03		8.39	11.0		192	250		15.0		14.4
Roach	K	9	0	0		2.19	1.82		0.90	0.62		0	0		0	0		4.37		4.03
	P	10	4.67	5.29		6.41	3.42		4.84	3.07		8.65	9.91		6.87	9.43		11.9		8.30
	V	10	55.9	53.4		11.5	8.26		12.8	6.60		46.5	52.0		20.0	15.3		23.8		20.0
Mussel	K	10	0	0		2.83	4.44		1.68	1.98		0	0		0.13	0.41		2.69		2.95
	P	9	1.44	2.21		7.44	3.47		5.62	3.81		13.2	23.3		4.79	4.88		5.09		7.33
Sponge	K	5	0.36	0.80		6.30	6.36		4.04	3.76		0.26	0.58		0.42	0.94		7.30		9.84
	P	5	6.86	9.14		1.45	2.43		13.0	14.7		1.34	3.00		11.1	20.6		7.24		6.18
	V	5	4.96	3.73		2.56	1.12		1.86	1.95		5.60	6.55		4.94	4.93		6.84		4.75
Plankton 100 m	K	4	0	0		0	0		20.6	28.0		0	0		0	0		0		0
	P	4	0	0		9.28	10.0		17.7	23.1		1.25	2.50		14.4	19.6		4.10		7.01
	V	4	2.45	4.90		9.90	1.73		6.55	3.51		8.85	6.05		5.83	2.40		78.5		45.3
Plankton 25 m	K	4	0	0		7.95	15.6		40.4	63.0		0.1	0.2		0.03	0.05		15.1		21.5
	P	4	0	0		14.3	10.5		9.95	5.89		0	0		0	0		8.58		9.55
	V	3	0	0		23.1	4.08		17.7	6.00		41.1	51.1		10.3	16.0		108		112
Sediment 0-2 cm	K	5	4.68	10.4		33.4	38.6		11.2	9.50		0	0		0	0		3.22		5.63
	P	5	10.7	15.1		37.5	29.3		5.34	3.80		2.68	3.68		6.42	5.30		13.9		22.0
	V	5	27.7	17.2		50.1	17.5		9.48	3.02		36.7	35.3		50.4	17.0		348		572

\* Trichlorophenol, tetrachlorophenol, pentachlorophenol, trichloroguaiacol, tetrachloroguaiacol, and tetrachlorocatechol.

\*\* K, P, and V are the lake sampling stations at Konnevesi, Paijanne, and Vatia, respectively.

† From Paasivirta et al. (1980). Reprinted with permission from Chemosphere, Vol 9, J. Paasivirta, J. Sarkka, T. Leskijärvi, and A. Roos, Transportation and Enrichment of Chlorinated Phenolic Compounds in Different Aquatic Food Chains, Copyright 1980--Pergamon Press, Ltd. Values are means  $\pm$  standard deviations expressed as parts per billion wet weight, except for sediments which are given as parts per billion dry weight.

Table 67  
Bioaccumulation of Di-2-Ethylhexyl Phthalate (DEHP), 1,2,  
4-trichlorobenzene (TCB), and Leptophos by Bluegill  
(*Lepomis macrochirus*) During Continuous Aqueous and  
Dietary + Aqueous Exposure\*

Day	Residue Body Burden, mg/kg**		
	DEHP	TCB	Leptophos
<u>Aqueous Only†</u>			
1	0.32(0.02)	0.29(0.13)	0.007(0.002)
3	0.55(0.11)	0.45(0.32)	0.045(0.011)
7	0.72(0.09)	0.82(0.41)	0.083(0.012)
10	0.52(0.09)	0.22(0.50)	0.149(0.052)
14	0.55(0.08)	0.79(0.42)	0.182(0.021)
21	0.83(0.16)	0.32(0.16)	0.150(0.053)
28	0.62(0.14)	0.57(0.10)	0.195(0.078)
35	0.66(0.13)	--††	0.195(0.050)
$\bar{X}$ ‡	0.64(0.11)	0.53(0.25)	0.179(0.03)
<u>Dietary + Aqueous‡‡</u>			
1	0.31(0.02)	0.23(0.11)	0.012(0.007)
3	0.57(0.09)	0.50(0.33)	0.037(0.007)
7	0.75(0.06)	0.33(0.20)	0.076(0.016)
10	0.64(0.10)	0.86(0.54)	0.141(0.031)
14	0.70(0.16)	0.78(0.38)	0.200(0.032)
21	0.90(0.35)	0.39(0.13)	0.175(0.017)
28	0.78(0.17)	0.55(0.09)	0.164(0.036)
35	0.78(0.27)	--	--
$\bar{X}$	0.73(0.11)	0.57(0.21)	0.170(0.030)

\* After Macek, Petrocelli, and Sleight (1979). Reprinted with permission from the American Society for Testing and Materials, 1916 Race Street, Philadelphia, PA 19103. Values are means with standard deviations shown in parentheses.

\*\* Body burden is based on the analysis of the whole fish for di-2-ethylhexyl phthalate and trichlorobenzene and on the analysis of the eviscerated carcass for leptophos.

† Aqueous exposure alone to 5.7 ppb DEHP, 2.9 ppb TCB, and 0.24 ppb leptophos.

†† The exposure was terminated after 28 days of exposure.

‡ Mean equilibrium body burden.

‡‡ Dietary + aqueous exposure: DEHP at 5.6 ppb aqueous + 2.8 ppm in diet; TCB at 2.9 ppb aqueous + 0.44 ppm in diet; leptophos at 0.22 ppb aqueous + 0.098 ppm in diet.



Table 68  
Concentrations of Dimethylnaphthalene (DMN) in Artemia sp. and  
P. pugio from the Six Exposure and Recovery Periods\*

<u>Temperature Regime</u>	<u>Palaemonetes</u> <u>pugio**</u>	<u>Artemia sp.†</u>
	<u>Exposure</u>	
Stable Temperature	5.26 ± 1.37 n = 20	0.24 ± 0.06 n = 9
Fluctuating Temperature	7.20 ± ±1.00 n = 20	0.24 ± 0.06 n = 9
	<u>Recovery</u>	
Stable Temperature	1.27 ± 0.35 n = 27	0.00††
Fluctuating Temperature	2.60 ± 0.48 n = 27	0.00††

\* From Dillon (1982). Reprinted with permission from the Bulletin of Environmental Contamination and Toxicology, Vol 28, p 152, Copyright 1982--Springer-Verlag. Values are means ± standard errors expressed as micrograms DMN per gram wet weight.

\*\* Each sample analyzed contained two shrimp.

† Each sample analyzed contained one food cube.

†† Uncontaminated food source.

Table 69  
Selected PAHs and Total PAHs in Flesh of Various Fish and Shellfish  
from the Hudson River and New York Bight Region\*

Species (location)	Naphthalene	Phenanthrene	Anthracene	Biphenyl	Total PAH
Atlantic mackerel ( <i>Scomber scombrus</i> ) (N.Y. Bight Apex)	ND**	10	ND	ND	10
Winter flounder ( <i>Pseudopleuronectes</i> <i>americanus</i> ) (Christiaensen Basin)	2	ND	ND	6	8
Winter flounder ( <i>P. americanus</i> ) (Raritan Bay)	2	1	ND	ND	5
Striped bass ( <i>Morone saxatilis</i> ) (Montauk Point)	7	ND	ND	ND	19
Striped bass ( <i>M. saxatilis</i> ) (Hudson River)	4	ND	ND	4	8
Lobster ( <i>Homarus americanus</i> ) (New York Bight)	7	ND	ND	ND	7
Lobster ( <i>H. americanus</i> ) (Raritan Bay)	5	5	ND	ND	25
Lobster ( <i>H. americanus</i> ) (Raritan Bay)	7	ND	ND	ND	77
Blue mussel ( <i>Mytilus edulis</i> ) (Sandy Hook)	6	6	ND	4	250
Blue mussel ( <i>M. edulis</i> ) (Shark River)	20	10	1	40	120

\* From O'Connor, Klotz, and Kneip (1982). Data are expressed as parts per billion wet weight.

\*\* ND indicates compound not detected.